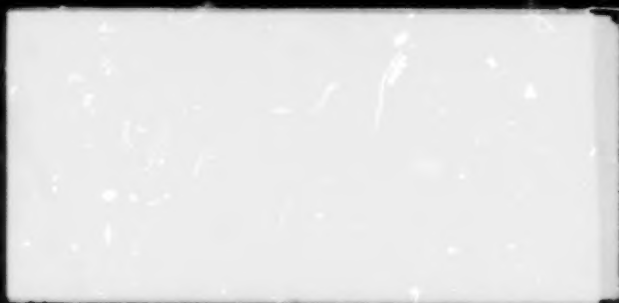


NATIONAL
URANIUM
TAILINGS
PROGRAM



PEER REVIEW OF THE NATIONAL URANIUM
TAILINGS PROGRAM'S (NUTP)
PROBABILISTIC SYSTEM MODEL
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Abstract

This report describes a peer review of the phase III component models developed by the National Uranium Tailings Program. The models are contained in the Uranium Tailings Assessment Program (UTAP)¹, which has been developed for long-term assessment of the environmental impact of a Canadian uranium mill tailings site. This critical review is intended to give added confidence in the applicability of the assessment models, and to suggest improvements where warranted.

The UTAP model considers a human receptor that neither inhabits the tailings itself, nor practices large-scale agriculture. Considering these limitations, the UTAP model, as a whole, is an acceptable assessment model. This review suggests minor improvements to the UTAP model, many of which should not change the model predictions significantly. In addition, the review describes areas of insufficient documentation supporting the UTAP model.

NOTICE

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1/ SENES. 1986. Probabilistic Model Development for the Assessment of the Long-term Effects of Uranium Mill Tailings in Canada - Phase III. Volume 3 - Uranium Tailings Assessment Program (UTAP) Component Model Documentation. A research report prepared under contract OAQ85-00182 for the National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, Energy, Mines and Resources Canada, 555 Booth Street, Ottawa, Ontario, K1A 0G1.

Executive Summary

The National Uranium Tailings Program (NUTP) has developed, under contract, a mathematical model for use in the Uranium Tailings Assessment Program (UTAP). This model is used to predict the long-term environmental effects of uranium tailings.

This report presents a peer review of the UTAP model as part of the quality assurance of the program. It was carried out by an independent group of scientific specialists, employed by Atomic Energy of Canada limited Research Company, to gain confidence in the mathematical models used in the UTAP code.

Our review is intended to either support the models, or to suggest improvements where there are inadequacies. We examined each component model in detail, giving support, criticisms, and recommendations. We have offered simple solutions where changes are recommended. In addition, we have made some general observations concerning the overall suitability of the UTAP model to predict the long-term consequences of uranium tailings.

The UTAP model applies to a human receptor that neither inhabits the tailings itself nor practises large-scale agriculture. We have considered this during the review, and have restricted our comments to the specified scenario. In general, the component models are, for the most part, technically adequate for the scenario considered. The documentation is often lacking in sufficient detail to be convincing, although the implementation of the component model is usually correct. We have suggested minor revisions to all of the component models, and most of our recommendations would not likely increase the predicted final radiation dose to man drastically.

We feel that further work is required to validate the UTAP model. An effective approach would be to compare predicted results with observed field data, and we recommend that this task should have a high priority.

In general, we also feel that the UTAP approach and model represent a significant advancement towards the long-term environmental assessment of uranium mill tailings.

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1. Introduction

1.1 Overview

The National Uranium Tailings Program (NUTP) was activated, under the direction of the Canada Centre for Mineral and Energy Technology, Energy, Mines and Resources Canada, to further research efforts on uranium mill tailings in Canada. One of the objectives of NUTP was to develop an approach to the long-term assessment of these tailings. The approach chosen was based on the probabilistic systems assessment methodology, and NUTP developed the Uranium Tailings Assessment Program (UTAP). UTAP was developed in three distinct phases with each succeeding phase having more detailed models. The Phase III version of UTAP contains a set of mathematical models that represent various components of the tailings system (SENES 1986).

This present report documents a peer review of the Phase III component models in UTAP, as part of the quality assurance process. Peer review by a group of scientific specialists, not directly connected with the development of UTAP, will provide additional support and confidence in the models embodied within UTAP. Other quality assurance exercises have involved tests of the UTAP code (SENES 1987) and an intercomparison with another probabilistic code (Goodwin and Andres 1986; Goodwin et al. 1987).

This review provides a detailed critical examination of all component models used in the Phase III version of UTAP. Where possible, a model is acknowledged as being acceptable and suitable for its intended use within UTAP. Otherwise, any proven or potential inadequacies are noted, and recommendations are made for model improvement. In addition, the documented data used by the models in UTAP have also been considered, and a critical review of the suitability of the data is provided.

1.2 Scope of Work and Personnel

The objective of this project is to prepare and document a peer review of the Phase III component models used in UTAP. The computer code was not examined. Rather, this review pertains only to the documentation of the models as prepared by SENES Consultants Ltd. (SENES 1986; Vol. 3).

Documentation for each major component model, and other related material, were provided to each of the members of the review team. Each topic was matched with the expertise of the reviewers, who then carried out an independent examination of the documentation.

Their task was to critically review conceptual and mathematical aspects of the models. Each reviewer also considered the applicability of parameters, and the corresponding parameter values and/or parameter probability distributions. Site-specific parameter data, such as exposed tailings surface area and watershed dimensions, were not reviewed since these parameters are fixed by physical constants imposed by the particular tailings location under study.

Questions concerning the models and their parameters were noted. We have also provided recommendation for improvements to the models and data when deficiencies were found. In some instances, we provide direction to literature sources for subsequent follow-up. The entire review team also met for a group review of all critical comments, and to examine areas of overlap such as the interfaces between the source and transport models.

The review team mentioned previously consisted of ten principal scientists and two consulting scientists from the Atomic Energy of Canada Research Company. Table 1.1 lists these scientists, together with their disciplines and/or areas of expertise.

Table 1.1
Principal Reviewers

B.D. Amiro, Ph.D.	Meteorology/Atmospheric transport of radionuclides
J.W. Barnard, Ph.D.	Health Physics/Dosimetry
R.J. Cornett, Ph.D.	Limnology/Aquatic transport of radionuclides
B.W. Goodwin, Ph.D.	Theoretical Chemistry/Chemical modelling, systems analysis
T.W. Melnyk, Ph.D.	Theoretical Chemistry/Groundwater transport of chemicals
S. Mihok, Ph.D.	Ecology/Food chain modelling
M.I. Sheppard, Ph.D.	Soil Science/Soil transport and chemistry
S.C. Sheppard, Ph.D.	Plant Science/Chemical uptake by plants
P. Vilks, Ph.D.	Geochemistry/colloids
R. Zach, Ph.D.	Ecology/Food chain modelling, dosimetry

Consulting Scientists

P.A. Davis, Ph.D.	Atmospheric physics
R.J. Lemire, Ph.D.	Chemistry

1.3 Review format

This report reviews each component model separately. The individual reviews are organized consistently with the following format:

1. Overview and description of requirements of component model
2. Comparison of model with requirements
3. Specific comments on model
4. Specific comments on parameters
5. Recommendations
6. Summary
7. References

Section 1 (Overview and description) provides a statement of the requirements that the component model should address. At this point, we accept that a compartment has been defined (e.g. soil), and discuss the dynamics of radionuclide movement within the compartment. A brief overview is also presented to describe concepts and to tie the individual component model to input and output parameters.

The second section (Comparison of model with requirements) is a critical review of the pathways selected to fulfill the requirements of the assessment. Interface of the component model with other models is also reviewed here.

The third section (Specific comments on model) reviews both the technical aspects of the model, and the supporting model documentation. The technical aspects are stressed and the logical development of the model is scrutinized. Comments of an editorial nature are also recorded. Many of the editorial comments do not affect the correctness of the model, but are included as constructive criticisms of the documentation. Areas of insufficient documentation are also noted.

The fourth section (Specific comments on parameters) concentrates on the selection of parameter values. In this section, we not only check for the use of reasonable parameter values, but employ the principle that conservative values should be used when there is a large degree of uncertainty. The UTAP model is probabilistic and our review of parameter values is based on the use of probability distribution functions.

Section five (Recommendations) lists our major recommendations for each of the component models. We made many points in the preceding sections, and only extracted those recommendations that provide a major benefit to UTAP. Some of our recommendations are concrete; for example, changes to parameter values. Other recommendations are more general, such as suggestions concerning the need for further model documentation.

Section six (Summary) summarizes the review of the component model. The summary is intended to be balanced, giving a brief account of areas of support, and areas where additional work is warranted. The references used in the component model review are listed in section seven.

We have also provided a concise overview of our findings in Chapter 16 summarizing the most important points of the component model reviews.

The component model documentation is arranged as separate sections (SENES 1986). We have followed these divisions, and reviewed the component models as individual tasks. Our review personnel were responsible for the individual sections listed in Table 1.2 (this report).

In our review, references to page numbers, equations, tables and figures normally correspond to those in the SENES documentation. References to equations, tables and figures reported in our review are labelled as such in our text. Parameter symbols are consistent between our review and the SENES documentation.

Table 1.2
Responsibilities of Review Personnel

SENES document section	Component model	Section in this review	Reviewer
3 (3.1 to 3.13)	Water source	3	P. Vilks
3.14	Tailings vegetation and erosion	4	M.I. Sheppard
4	Air source	5	B.D. Amiro
5	Air dispersion	6	B.D. Amiro
6	Soil	7	M.I. Sheppard
7	Groundshine	8	J.W. Barnard
8	Terrestrial vegetation	9	S.C. Sheppard
9	Subsurface transport	10	T.W. Melnyk
10	Surface water dispersion	11	R.J. Cornett
11	Aquatic	12	S. Mihok & R. Zach
12	Animal produce	13	S. Mihok & R. Zach
13	Dose	14	S. Mihok & R. Zach
Appendix C	Episodic events	15	B.W. Goodwin

1.4 References

- Goodwin, B.W. and T.H. Andres. 1986. Comparison of the probabilistic systems assessment codes, UTAP and SYVAC3, applied to the long-term assessment of uranium mill tailings in Canada. A research report prepared under contract #15SQ.23317-5-1720 for the National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, Energy, Mines and Resources Canada, 555 Booth Street, Ottawa, Ontario, K1A 0G1.
- Goodwin, B.W., T.H. Andres, R.J. Lemire and P. Ho. 1987. Comparison of the National Uranium Tailings Program's Probabilistic Code UTAP2 with Atomic Energy of Canada Limited's Probabilistic Code SYVAC-UT2. A research report prepared under contract #15SQ.23317-6-1731 for the National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, Energy Mines and Resources Canada, 555 Booth Street, Ottawa, Ontario, K1A 0G1.
- SENES. 1986. Probabilistic Model Development for the Assessment of the Long-term Effects of Uranium Mill Tailings in Canada - Phase III. Volume 3 - Uranium Tailings Assessment Program (UTAP) Component Model Documentation. A research report prepared under contract OAQ85-00182 for the National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, Energy, Mines and Resources Canada, 555 Booth Street, Ottawa, Ontario, K1A 0G1.
- SENES. 1987. Phase III software development and quality assurance supplementary report, in preparation.

2. General Comments and Overall Structure

The general objective of the UTAP model is to assess the long-term environmental effects of uranium mill tailings. The model consists of several component models, used to simulate the physical and chemical processes that operate in a uranium mill tailings environment, that may affect the nearby environment. The problem is analyzed probabilistically for a hypothetical reference site. The assessment considers a period of 1,000 years, covering a study area within a 50 km radius of the tailings. The major end product of the model is an estimated dose to a human receptor within this area. This may be either a "critical" or "average" individual located within a specified sector. A second major end product of the model is the estimated effect of the tailings on the pH of lakes and rivers downstream of the tailings site. The pH is of interest because of potential effects on the biota, and on drinking water quality.

We did not review the logic and underlying philosophy that contributed to the selection of the pathways included in UTAP. The available documentation does not justify inclusion or exclusion of pathways. We would have preferred that the documentation list all potentially important pathways, and then justify the process of selective inclusion. An alternative approach is to state explicitly the basic assumptions used to define the pathways selected for the UTAP model. Either of these two approaches is preferable to that of using undocumented and unstated assumptions.

From our review, we have determined two major assumptions dictating the pathway selection in UTAP:

- (1) There is no intrusion on the tailings by man; and
- (2) Large-scale agriculture is not practised within the study area.

All pathways exclude these two scenarios although they may be important when considering a long-term assessment extending to 1,000 years.

Nevertheless, the pathways outlined in Fig. 2.1 of the SENES report (SENES 1986) are a reasonable subset of those that should be considered in an environmental assessment of uranium mill tailings. Throughout the documentation, it would have been very helpful to have more flow diagrams describing pathways, with special detailing of interfaces between pathways. It proved difficult to track the transfer of radionuclides between some compartments, especially between the tailings water source and subsurface transport models. Flow diagrams, giving qualitative lists of elements within compartments, would be very helpful (for example, the list might show whether barium is transferred from the tailings source model to the subsurface transport model). Discussion of the transport of radionuclides between compartments and ingrowth of decay chains within compartments could be greatly improved.

A sensitivity analysis would help select pathways and parameters within each component model.

We also feel that the component models in UTAP could be more satisfactorily supported by comparison of predicted results with field observations. Although the models are long-term models, application of the code shows that many significant effects are expected within the first few years following the production of a tailings pile. It should be feasible, therefore to validate many aspects of the component models, at least on a time scale of tens of years. We recommend that future quality assurance procedures place a strong emphasis on validation through comparison of predicted results with field observations.

Despite the problems with the documentation of the model philosophy, most of the logic inherent in the UTAP model is reasonable. In our review, we have concentrated most of our specific comments on technical aspects of the modelling of the selected pathways. Our comments are based on the requirement for the development of an assessment model for the NUTP. Therefore, we did not dwell on the added accuracy which more detailed research models could add to short-term simulations of radionuclide transport, but we have reviewed UTAP in the context of the requirements of a long-term environmental assessment.

3. Review of Uranium Mill Tailings Water Source (P. Vilks)

3.1 Overview and description of requirements of component model

The uranium tailings are a source of calcium, iron, sulphate, acidity, and radionuclides, which may be transported to the environment. The source term model is critical to the entire assessment model because it provides the concentrations of radionuclides and other contaminants which leave the tailings via the groundwater and surface water, and it determines the radionuclide concentrations in the tailings' surface layers which are subject to wind erosion and dispersal.

In order to calculate the pore water chemistry of mill tailings as a function of depth and time, the following must be considered:

- percolation of rainwater downward through the tailings.
- change in redox conditions with depth.
- oxidation of pyrite.
- precipitation and dissolution of sulphates.
- mineral weathering.

The source model must identify the outputs to surface water, subsurface water, and the atmosphere.

The source term model should describe the pore water chemistry (sulphate, calcium, iron, pH, aluminum) as a function of time and depth in the tailings. The pore water chemistry is then used to define the radionuclide dissolution from the solid phases. In addition to dissolution reactions, dissolved radionuclide concentrations at different times and depths can be modified by radionuclide ingrowth and decay, rainwater percolation, and from precipitation and sorption processes. An inventory of solid-phase radionuclides is required as a function of time and depth. The dissolved concentrations are required as inputs to the subsurface transport model and seepage to the surface water dispersion model. The solid concentrations are required as input to the air dispersion model.

3.2 Comparison of model with requirements

The water source model divides the tailings into four geochemically distinct layers in order to deal with changing redox conditions and to define outputs to air, surface water, and groundwater. The upper layers (1 and 2) are assumed to be oxidizing, while layers 3 and 4 are reducing. Rainwater falling on the tailings percolates through the tailings, transporting ions downward from one layer to the next. In layer 1, sulphates have been leached out and accumulated in layer 2. Layer 3 is a mixing zone, where oxidized waters from the upper layers are reduced, and radionuclides can sorb onto solid particles because of the higher pH. Ions in layer 3 are considered to be the source to surface water (Figure 3.1, SENES report), while layer 4 is the source to groundwater. The inventory of solid phase radionuclides in layer 1 is a source to the air dispersion model.

The model assumes pyrite oxidation is only in the upper two layers, where it is kinetically controlled. The pyrite oxidation rate is assumed to be first order with respect to pyrite content. The release of acidity, sulphate, iron and uranium from pyrite oxidation has a significant impact on the pore water chemistry of the entire mill tailings pile.

The sulphate released from pyrite is assumed to precipitate gypsum and barite, which then buffer the concentrations of sulphate, calcium and barium. Barite is assumed to regulate the dissolved concentration of Ra-226 by coprecipitation of RaSO_4 with barite. The formation of anglesite (PbSO_4) controls the solution chemistry of Pb-210.

The acidity released from pyrite oxidation induces the weathering of primary and secondary minerals in the mill tailings. Mineral weathering affects the pore water pH by releasing ions to solution and by consuming hydrogen ions. The source model assumes that iron hydroxide, kaolinite, and amorphous silica are the important secondary minerals controlling the solubility of Fe(III), aluminum, and silica. The model acknowledges the presence of primary aluminosilicates such as feldspars. It is assumed that the weathering of these silicates is too slow to affect the pH of layers 1 and 2 due to the large amount of acidity released from pyrite. In layers 3 and 4, where silicate weathering may have a greater impact due to higher pH, the effect of aluminosilicates is modelled by albite weathering. This assumption seems to be an oversimplification. However, in light of the absence of direct acid neutralization capacity measurements on mill tailings, the model approach is adequate. In any case, it is a conservative assumption because it may underestimate pH. The buffering due to calcite is also considered if lime has been added to the tailings.

The estimation of radionuclide migration requires the model to calculate the mass balance of radionuclide inputs and outputs to the pore water in each layer. Once the pore water chemistry and downward percolation rate are known, the solid phase radionuclide inventory for each layer can be maintained as a function of time. The radionuclide inputs to the pore water chemistry which need to be considered are dissolution, ingrowth, and migration from an upper layer. Removal from solution can occur by radioactive decay, precipitation, sorption, or migration to another layer.

The inputs and outputs depend upon the type of radionuclide and the layer being considered. Table 3.1 (this report) summarizes the mechanisms used by the water source model to account for the geochemical behavior of radionuclides as a function of depth. Ingrowth and decay of radionuclides in the U-238 chain is straightforward. Radionuclide removal from solution by sorption is approximated by distribution coefficients. Precipitation of thorium is probably a conservative approximation because sorption reactions should remove more thorium from solution than predicted by solubility. Rn-222 is not included in the table but is part of the air source term model.

Table 3.1
Mass Balance of Radionuclides in Solution

Radionuclide	Layer	Source	Removal from solution
uranium	1,2	pyrite (as U_3O_8)	
	3,4	layers 1,2	sorption
thorium	1,2	thorium hydroxide	
	3,4	thorium hydroxide layers 1,2	precipitation
radium-226	1	ingrowth from Th-230	decay to Rn-222
	2	$BaSO_4$ ($RaSO_4$) ingrowth from Th-230	decay to Rn-222 coprecipitation with $BaSO_4$
	3,4	$BaSO_4$ ingrowth from Th-230 layer 2	decay to Rn-222 sorption (K_d based on coprecipitation with $BaSO_4$)
lead-210	1,2	$PbSO_4$ ingrowth from Ra-226	decay
	3,4	layers 1,2 $PbSO_4$ ingrowth from Ra-226	sorption decay

After each time step, an inventory is calculated of solid phase radionuclides, as well as certain key minerals including pyrite and the sulphates. As this inventory changes with depth and time, the source term model approximates the movement of radionuclide and mineral fronts downward through the tailings.

3.3 Specific comments on model

pg. 3-8 Equations 3.3.2, 3.3.3, and 3.3.4 appear to be from Nordstrom, 1982. In any case a reference is required.

pg. 3-9 In equation 3.3.10 the specific rate constant of pyrite oxidation is calculated from the initial concentration of pyrite. Will this constant change as the pyrite concentration decreases with time, and if so, is the specific rate constant recalculated after each time period?

pg. 3-21 As considered before, is the rate constant (k') for pyrite oxidation corrected for decreasing pyrite inventory with time?

- pg. 3-26 The equations 3.4.6 and 3.4.7, which are used to calculate dissolved calcium and sulphate in layer 3, do not contain a correction for sulphate complexation, as was found in 3.3.39 and 3.3.40. Is this an omission or is it assumed that Fe^{3+} and Al^{3+} will not be significant due to the lower oxidation state and higher pH?
- pg. 3-33 The assumption that basaluminite will not affect aluminum solubility may be optimistic. Dubrovsky et al. (1984) have found pH's low enough (0.95 to 1.5) in pyritic uranium tailings to make basaluminite a stable aluminum mineral. In the absence of basaluminite there may be other aluminum phases, such as allophane or jurbanite, which control Al solubility. However, in the absence of a clear consensus on which minerals determine aluminum solubility, kaolinite is a reasonable choice.
- pg. 3-38 (see also pg. 3-41 and pg. 3-47) The electroneutrality equations, which are used to calculate pH for layers 1 and 2, should also include $\text{Al}(\text{OH})_2^+$, in case the pH goes up to 4. The electroneutrality equation for layer 3 must definitely include $\text{Al}(\text{OH})_2^+$.
- pg. 3-39 The ion balance (equation 3.6.18) added to account for limestone addition is not derived and the origin of this equation is not explained or referenced. (This is annoying.)
- pg. 3-42 The derivation of the electroneutrality equation for the lime case should be made more clear or referenced. (Although the derivation is much more clear than for equation 3.6.18.)
- pg. 3-45 The calculation of sodium concentration (to represent the weathering of all alkalies from silicates) seems very simple, although it is probably sufficient. Is there a reference for this approach?
- pg. 3-46 The listing of sulphate species did not include PbSO_4 and BaSO_4 , which were mentioned for layer 2 (page 3-41). Although they do not significantly contribute to the sulphate species, they could be listed for consistency.
- pg. 3-47 Ba^{2+} and sulphate, derived from barite, are not included in the electroneutrality equations. Shouldn't they also be included for consistency, and to model the areas in layer 1 where barite is present and gypsum has just been depleted?
- pg. 3-54 The text indicates that some fraction of the uranium which was not extracted in the mill is associated with pyrite. The reader is left wondering where the rest of the uranium is. Perhaps the text should indicate the remaining uranium is in silicates and is assumed to be insoluble for the duration of the assessment.

pg. 3-67 to pg. 3-69

A brief discussion on the derivation of equation 3.8.4 (and 3.8.5) is warranted. Uranium removal at the bottom of each zone depends upon a balance between the flushing rate of rainwater and the interaction of dissolved uranium with solids. The mathematical procedure used to account for uranium flushing and retardation should be outlined or referenced. The following points concerning the uranium solid: liquid distribution ratio and solution chemistry should also be clarified or referenced:

- Is the K_d based on sorption or coprecipitation reactions?
- Is uranium solubility checked at any time?
- Uranium speciation is not even considered. Redox conditions and complexation with carbonate species have significant effects on uranium solubility. If uranium speciation is not considered to be important, a note should be made justifying this assumption.

pg. 3-69 It is assumed that the only source of uranium is from layer 2, because pyrite does not oxidize in layer 3. However, the source term model also assumes that Fe(III) which enters level 3 is reduced by pyrite. This must induce pyrite oxidation, followed by the release of uranium. Has this been considered (as a source of additional uranium, sulphate, and acidity)?

pg. 3-72 The statement is made that thorium solubility increases with depth in the unsaturated zone, and it may be useful to give the following reasons for this conclusion:

- 1) pH does not increase with depth in the unsaturated zone.
- 2) Sulphate complexation increases thorium solubility as sulphate concentration increases with depth.

pg. 3-86 There appears to be an error in equation 3.10.25. The equation is missing a term for the concentration of insoluble radium initially present in layer 2. (as was included for layer 1, equation 3.10.24).

pg. 3-87 Will not thorium leached from the unsaturated zone, which travels to the saturated zone and remains in solution, contribute to radium ingrowth? If so, why does the model distinguish radium ingrowth from thorium which has dissolved in the saturated zone from radium ingrowth from the total dissolved thorium.

pg. 3-88 Some explanation for the meaning of the radium solid-liquid distribution coefficient should be given or referenced (e.g. page 3-105). The text should clarify why a K_d approach is used for radium removal from solution in layers 3 and 4, while a coprecipitation calculation is used in layer 2.

pg. 3-93 The assumption that Pb-210 ingrowth can be calculated directly from the liquid phase concentration of Ra-226 may not be very good.

- 1) There are several isotopes between Ra-226 and Pb-210 in the uranium series decay chain, including Rn-222 (3.8 day half life). Since radon is a gas there is a possibility that it could migrate independently of Ra-226 before decaying to Pb-210.
- 2) Ra-226 in the solid phase will also decay to Rn-222, which will likely enter the liquid phase because it is a gas. Therefore, if Ra-226 is modeled as the direct parent of dissolved lead, the Ra-226 in solid form should also be considered.
- 3) Rn-222 could migrate from layer 2 and deposit extra Pb-210 on the surface layers.

In summary, a more rigorous calculation of Pb-210 ingrowth would make use of the Rn-222 concentration in the porewater instead of using dissolved radium. However, this comment is not meant to detract from the present model because the Rn-222 flux to layer 1 from layer 2 is not known. Perhaps future work on tailings piles could address the effect of upward Rn migration.

pg. 3-95 Why does the model set Pb-210 ingrowth to zero if Ra-226 is coming in from the layer above? Surely this Ra-226 can also decay to Pb-210. Perhaps Rn-222 flux is a better parameter to calculate Pb-210 ingrowth. Again an explanation could be given or referenced for the K_d term in equation 3.11.13.

3.4 Specific comments on parameters

- pg. 3-7 The formula for gypsum is $\text{CaSO}_4 \cdot \text{H}_2\text{O}$, not CaSO_4 , which is anhydrite. This really does not have an impact on any of the calculations, but it could cause the reader to wonder if the solubility data used in the model were for gypsum or anhydrite.
- pg. 3-17 The derivation of the factor d, which is used to calculate the uncomplexed sulphate should be shown or referenced.
- pg. 3-20 A reference is required for the assumption that carbonate buffered pH's will reduce pyrite oxidation by 1/10 to 1/20, since 1/20 is used in the calculations.
- pg. 3-36 The assumption that the thermodynamic data for iron species is based on concentration units and not activity should be referenced. All other species are corrected for ionic strength.
- pg. 3-37 I would like to see a reference for the carbonate constants at 10°C. Why are carbonate equilibria calculated for 10°C, while the other stability constants are for 25°C ?
- pg. 3-43 There is an error in the equation for $\text{Al}(\text{OH})^{2+}$. The constant 1.53 should be 1.83 to account for the lower Si activity in layer 2. Fortunately this mistake is not carried over to the electroneutrality equation for layer 3.

- pg. 3-82 The solubility products for radium sulphate and barium sulphate are not correct and do not match table 3.4. The solubility product of radium sulphate is not in table 3.4.
- pg. 3-108 The approximations used to calculate the upper values for the Ra-226 K_d are not referenced and do not appear to be justified. A conservative approach would be to use the lowest value for the K_d (5 ml/g), which has been referenced. The lower K_d would increase the concentration of Ra-226 leaving layer 4.
- pg. 3-110 Are the distribution coefficients used in the source term consistent with distribution coefficients used in the other models (such as the Subsurface Transport Model)?

3.5 Recommendations

- a. Include $Al(OH)_3^+$, Ba^{2+} , and the hydrogen ion in all electroneutrality equations which are used to calculate pH.
- b. The model should include uranium speciation calculations to check that uranium solubility has not been exceeded. Otherwise, some justification should be given why it is possible to ignore uranium solubility.
- c. The distribution of K_d 's for Ra-226 and Pb-210 should rely more on measured K_d values or apparent K_d observed in acid soils. Ideally the K_d distributions should also reflect the effect of pH on radium and lead sorption.
- d. Clarify some of the documentation describing the water source model, as mentioned in the specific comments.
- e. The model assumes certain key minerals and ions control water composition and pH. It would be very useful if the text would list the key minerals and elements in one location. The text should also mention other elements and minerals which are present in uranium tailings, and which are considered not important for controlling water chemistry.

3.6 Summary

- a. In many cases the model concepts are not well documented or the explanation is poorly organized. For example, concepts which are required for the calculation of ions in layers 1 or 2, are found in the section on pH.
- b. The overall concept of the model is appropriate for uranium mill tailings. Mill tailings, containing oxidizing pyrite, are a complex geochemical system. Many simplifying approximations had to be taken to model this system. However, these assumptions were not unreasonable for an assessment model.

- c. Some improvements could be made in the probability distribution functions of certain parameters, such as:
 - radionuclide K_d 's
 - rate of pyrite oxidation at higher pH
 - rate of albite weathering
- d. An attempt should be made to justify the water source term model by comparison of model predictions with field data.

3.7 References

- Dubrovsky, N.M.; K.A. Morin, J.A. Cherry and D.J.A. Smyth, 1984. Uranium tailings acidification and subsurface contaminant migration in a sand aquifer. *Water Poll. Res. J. Canada*. 19:55-89.
- Nordstrom, D.K. 1982. Aqueous pyrite oxidation and the consequent formation of secondary iron minerals. In: *Acid Sulphate Weathering*, Soil Science Society of America, pp. 37-55.
- Veska, E. 1983. Origin and subsurface migration of radionuclides from waste rock at an abandoned uranium mine near Bancroft, Ontario. Ph.D. Thesis, University of Waterloo.

4. Review of Tailings Erosion Model (M.I. Sheppard)

4.1 Overview and description of requirements of component model

Most of the requirements of the erosion model are met by the use of the Universal Soil Loss Equation (USLE). The USLE calculates soil loss from the tailings surface by water erosion. Several site-specific parameters are required for the equation such as size and shape of tailings area, slope, soil particle size, soil moisture content, rainfall and presence or absence of vegetation. The soil lost from the tailings could result in a downstream deposition of soil enriched with radionuclides, as well as a radionuclide concentration input to a stream. This model could also be applied downwind where the dust model deposits its load since erosion occurs on any bare soil surface.

4.2 Comparison of model with requirements

The USLE calculates the loss of soil from the tailings, yet no discussion exists as to how this loss is then input to determine the downstream or downwind soil or water concentrations. The use of the USLE ends abruptly as soon as the tailings are depleted. The model used is appropriate for calculating soil loss, incorporating a version developed by Haan (1978) to partition the gross erosion into bedload and suspended sediment load.

The model is appropriate but not put into context. How important is the loss of 3.5 cm of tailings over 1,000 years compared with some other process? It is likely insufficient to include floods as only a rare episodic event (Section 15, this review). Floods should be modelled explicitly.

4.3 Specific comments on model

It is not stated clearly how bedload transport and suspended sediment transport capacity account for deposition. A discussion of the difference between bedload and suspended sediment load should be given, including demonstration of what particle sizes or aggregate sizes are affected by each process. This is not explained fully enough on pg. 3-118.

The tailings model includes another model, a revegetation model, which is defined as a simple exponential tailings vegetation model. What does this mean, exponential in what regard? Equations 3.14.1 and 3.14.2 are poorly coupled; the development of how 3.14.2 is derived from 3.14.1 must be shown. Many terms are not defined in both of these models such as soil structure code, soil permeability class and re-vegetation half-life. Figure 3.12 describes equation 3.14.6, but $C_s = 1$ at $F_v = 0$ is not consistent with either the figure or the equation (this boundary condition is not clear). The original reference does not list equation 3.14.6; this equation must be referenced.

There are several inconsistencies between the text and the code. For example, slope is defined as ≤ 2.5 or 3.0%, and silt is assumed to be

≤ 0.02 or ≤ 0.06 mm. The term "fraction of activity which instantly dissolves", on the bottom of pg. 3-130 is not defined. How is this fraction determined; do they use K_d ? If so, then a reference is required.

The final erosion rate calculated should have been compared to existing data, such as for silt soils in dry locations. The final erosion rate is given, but no scenario is described for this rate to be put into perspective. Information like rainfall and particle size are all hidden in erosion indices and permeability factors, and the real scenario is hidden from the reviewer.

4.4 Specific comments on parameters

The site-dependent parameters such as slope, shape and permeability are not justified with respect to a real tailings area. Some terms are not defined correctly or explicitly, such as a slope of 400'. This is the slope length. It is confusing whether slope lengths are segments from the top of the tailings pile, or exactly what lengths are used in equation 3.14.2.

The SENES report departs from using a traditional value for C_s of 0.45 for $F_v = 0$ to assume a $C_s = 1$ for $F_v = 0$ (pg. 3-122). This change is reasonable, but the justification given for this new value is not very clear, and there is no attempt to show that it makes any difference to change this value. There is no boundary condition given for T in equation 3.14.7. The bedload transport model (equation 3.14.8) has no units stipulated for its variables, hence the form of the equation cannot be checked. What is depth of flow (d_f) in equation 3.14.15 of the suspended sediment transport model? What is V , the flow velocity of what through the grass? No conversion is ever shown to convert seconds in equations 3.14.15 and 3.14.16 to years. S_s is given in feet on pg. 3-128 and centimeters on pg. 3-129. In equation 3.14.20, is D_i really D_{50} ? Is tailings bulk density in Table 3.6 a wet or dry bulk density?

The probabilistic parameter values are not justified, nor is the need to have parameter values probabilistic justified in this model.

4.5 Recommendations

- a. Explain how using bedload and suspended sediment transport capacity accounts for deposition and show where the deposition information is used.
- b. Model the runoff water concentration and use it as a stream input. This is done simply through infiltration rate, surface area and a dissolution term.
- c. Justify the use of the USLE and define the exact scenario pertinent to a real tailings site.
- d. Compare water erosion consequences, using the USLE, to sheetwash from a 1 in 1000 year flood. This was not done adequately in the appendix (also see section 15, this review).

- e. Justify the probabilistic parameter values.

4.6

Summary

- a. The USLE appears to be appropriate for assessing soil erosion on the tailings.
- b. Documentation, assumptions, parameter values and applicability of the model require further justification.

4.7

References

Haan, B. 1978. Hydrology and Sedimentology of Surface Mined Lands. Office of Continuing Education and Extension, College of Engineering, University of Kentucky, Lexington, Kentucky.

5. Review of Air Source Model (B.D. Amiro)

5.1 Overview and description of requirements of component model

The assessment requires an estimate of the flux of five elements (natural uranium, total thorium, Ra-226, Rn-222, Pb-210) from the tailings to the atmosphere. This flux is used as a source term in a subsequent model of atmospheric dispersion to calculate air concentration. Radon is the only volatile element that might leave the tailings area as a gas. Other elements must be attached to suspended particulate matter.

5.2 Comparison of model with requirements

The air source model estimates both gaseous radon emission, and contaminated dust suspension. Both of these suspension mechanisms are modelled using currently-accepted concepts. There is no discussion to support the neglect of other mechanisms. It is assumed correctly that vegetation will reduce soil erosion by wind. However, contaminated vegetation can be burned by natural (forest or grass) fires or by anthropogenically controlled (agricultural) fires. These fires can suspend contaminated particulate matter and may be more important than dust suspension. Support for neglecting dust suspension by cultural activities (e.g. farming) is also needed.

5.3 Specific comments on model

The radon source term is based on a model by Rogers et al. (1980). This is a well-accepted model and is satisfactory for the requirements. Some assumptions are made to simplify the model. The following assumptions are reasonable:

- a) The unsaturated zone is the only source region.
- b) Air concentration of radon at air/tailings interface equals zero.
- c) The moisture and porosity of the two unsaturated-zone layers are identical.
- d) Vegetation has no effect on radon emission.

The derivation of the equations is correct and the empirical form for the diffusivity of radon is valid. The description of the model equations is vague in the text, but the code used equations 4.2 and 4.4. Assumptions about annual averages, and lower emissions during the winter months are satisfactory.

The dust source model gives a crude estimate of wind erosion. The model used is superior to a mass-loading approach and comparable to the wind-erosion equation (described by Male 1985). It is likely not as good as mechanistic models (see review by Smith 1983) but the assessment does not warrant the added complexity. The model used is satisfactory and its implementation appears to be correct.

Vegetation is assumed to reduce dust flux and changes in the erodible fraction of soil with time are ignored. No dust suspension is assumed during the winter months. These assumptions are reasonable. Time averaging throughout the year is a questionable practice, depending on the technique of incorporating dispersion. This problem is addressed in section 6 concerning the air dispersion model.

5.4 Specific comments on parameters.

a) Radon source parameters

The site-specific parameters (refer to SENES report for parameter definition), A_s , M , E , and N_w were not reviewed but they appear to be reasonable. The thickness of the unsaturated layer, $x_u(t)$, should have a distribution of $U(0.75, 1.25)$ to correspond to both its nominal value and to λ_u , which is the same parameter in the tailings water source model.

The calculated parameters a , D , $J_T(t)$, $x_1(t)$, $x_2(t)$ and $Ra_1(t)$ are satisfactory as inputs and outputs. The constant λ is correct. E is a generic parameter for which we only have limited information. The values given for E are satisfactory but E really has units of $Bq\ Rn \cdot Bq^{-1}\ Ra$.

The value of the winter/summer radon flux ratio, R , was not selected properly from the data of Zettwoog et al. (1981). Although data are sparse, it would be best to choose R -values typical of the moisture content in the tailings, with a similar triangular distribution. We would suggest a distribution of $T(0.0025, 0.3, 0.4)$ which is both more realistic and conservative than the distribution indicated in Table 4.1.

b) Dust source parameters

The site dependent variables s , P_s , f , ρ and N_w were not reviewed but appear to be reasonable. The parameters $C_i(t)$ and $A(t)$ are calculated outside of the air source model. $Q_{1\ AVG}(t)$ is applicable if the dispersion term is modified to only consider cases when wind erosion does occur (see section 6.3, this review).

5.5 Recommendations

- a. Suspension of all elements from vegetated tailings during a fire event should be considered. Perhaps an addendum could justify the exclusion of this pathway, but it is possible that it will be more important than dust suspension, at least over 100's of years. Two scenarios can be considered: 1. a forest fire about every 100 years, and 2. a more frequent grass fire. Forest fires are mentioned in the appendix as only contributing to enhanced wind erosion.

For the fire scenario, one could assume that all radon incorporated in the vegetation is released. Other elements would be released attached to particulates. Their suspension can be estimated using particulate emission

factors. For example, a forest fire emits 75 g particulates·kg⁻¹ fuel and a fire every century with 2.2 kg fuel·m⁻² gives about 16.5 kg particulates·ha⁻¹·a⁻¹ (Amiro 1985).

- b. The distribution of R should be changed to T(0.0025, 0.3, 0.4), and $x_s(t)$ to U(0.75, 1.25).

5.6 Summary

- a. The air source models appear to be satisfactory in their application to the UTAP model.
- b. Justification is needed to neglect radionuclide suspension by other methods (e.g. fires).
- c. The parameter distributions of R and $x_s(t)$ are not satisfactory and recommended values are given.

5.7 References

- Amiro, B.D. 1985. Suspension of radionuclides into the atmosphere in the vicinity of an underground nuclear waste vault. Atomic Energy of Canada Limited Technical Record, TR-366. Pinawa, Manitoba.
- Male, D.H. 1985. Wind transport of soil aerosols. Atomic Energy of Canada Limited Technical Record, TR-295, Pinawa, Manitoba.
- Rogers, V.C., R.F. Overmyer, K.M. Putzeg, C.M. Jensen, K.K. Nielson, and B.W. Sermon. 1980. Characterization of uranium tailings cover materials for radon flux reduction. U.S. Nuclear Regulatory Commission, NUREG/CR-1081.
- Smith, W.J. 1983. An evaluation of models of particulate suspension for a thorium ore stockpile. Colorado State University, Ph.D. thesis. Fort Collins, Co.
- Zettwoog, P., N. Fourcade, F. Campbell, and H.S. Caplan. 1981. The effect of freezing uranium ore and tailings samples. Health Physics 41:410-411.

6. Review of Air Dispersion Model (B.D. Amiro)

6.1 Overview and description of requirements of component model

An air dispersion model is required to calculate an air concentration of radionuclides resulting from a flux from the tailings surface. The dispersion model must be able to predict concentrations for both gases and particles at the receptor height (generally between 1 and 2 m). Air concentrations are required as a function of distance from the source, and time-averaged site-specific wind direction. The concentrations are used as inputs to inhalation and immersion pathways, and to deposition on vegetation and soil.

6.2 Comparison of model with requirements.

The air dispersion model calculates air concentrations downwind of the tailings using flux from the tailings as an input. The model is a Gaussian point-source model, modified to calculate sector-averaged concentrations. It also makes assumptions to convert a point-source model to an area-source model.

The model can work conceptually for a receptor not located on the tailings, and gets better with greater downwind distances. It cannot calculate air concentrations over the tailings source. Therefore, in principle, it is suitable for sector averaging for the outer rings of the reference site, but it is not reasonable if the inner ring includes only the tailings area itself.

6.3 Specific comments on model

A Gaussian dispersion model is quite standard, although this type of model is awkward for area sources. Parts of the model are very poorly documented. It is not clear from which point the distance x is measured (centre of tailings?). The sector-averaging assumptions are not stated and a reference is only given to phase I documentation, which itself is not clear. Although the constant 2.032 was not explained, it is correct. The mean wind speed, u , given in equation 5.1 should be u_{K1} and depends on stability class.

The virtual source correction attempts to convert the basic point-source model to an area-source model. The effective circular area source assumption is satisfactory and the geometry assumptions are reasonable, although inadequate documentation is given. However, it would make good sense to use the effective non-vegetated tailings area (AEXPT) as A_0 (parameter symbols are those used by SENES) in equation 5.5 for all elements except radon. This is because the source area, for the elements suspended with DUST, decreases with vegetation cover, and although AEXPT was used in the DUST source model, it should also be used for dispersion. The present use of $A_0 = \text{ATAIL}$ is not conservative because it gives a larger value for x_0 , that in turn, gives a smaller value for the air concentration, $C_{ai}(t)$. ATAIL should be maintained for the dispersion of radon because vegetation is not assumed to alter the radon source term. The

virtual distance accounts for dispersion in the y dimension (e.g. Turner 1970) and letting $L = D$ ensures that the sector plume is close to the correct width. However, σ_z should not be dependent on the virtual source distance, but is only a function of the actual distance from the real source (i.e. $\sigma_z = a(x)^b$; equation 5.6). The assumption for σ_z used in the present model will give a concentration that is too low and is unrealistic. To be conservative, the x used for σ_z should be measured from the edge of the tailings closest to the receptor.

The model is not appropriate to calculate air concentrations on the tailings itself, or close to the edge. Table 5.2 gives a minimum distance of x of 500 m. This is a reasonable distance from the edge of the tailings where the Gaussian model can start to be applied. Hence, if the receptor is closer than 500 m from the tailings, a different model should be used.

A source height of 2 m is assumed, but a ground-level height could have been chosen where $h = 0$. This is not critical since the exponential term is inconsequential for most of the distances modelled. The power law coefficient concept, used to calculate σ_z , conforms with currently-accepted models. Concentration averaging within the rings and area-weighted averages for browsing animals are reasonable.

The wind-speed classes and conceptual use of STAR data are satisfactory for radon dispersion because the radon flux is assumed to be steady-state. However, dust is only generated when $\bar{u} > 5.4 \text{ m}\cdot\text{s}^{-1}$, and dispersion should only reflect these cases. The model presently uses meteorological data representative of long-term averages. It is not easy to determine whether this is conservative because both $\bar{u}$ and σ_z must be considered. This problem should at least be qualified. It would be easy to modify the model inputs to only include wind-speed frequencies for classes ≥ 3 (to be conservative) and use the σ_z function for class D stability (most winds $> 5.4 \text{ m}\cdot\text{s}^{-1}$ will be C or D, and D is conservative). This method would assume that precipitation events are distributed equally across wind-speed and stability classes.

Another implicit assumption is that the frequency of meteorological conditions are distributed similarly throughout the year (i.e. wind-speed, wind-direction and stability statistics are similar for both summer and winter conditions). This assumption is necessary because the STAR data statistics are not time dependent whereas the radionuclide air source ($Q_i(t)$) does not occur in winter. For long-term assessment, this assumption is likely satisfactory. Documentation on the method for implementing the STAR data is lacking in the text. It would be difficult to implement this model while performing an assessment based on the information available.

6.4 Specific comments on parameters

The site-dependent variables A_o , f_{k1} , and x were not reviewed. The calculated variables $C_{ai}(t)$ and $Q_i(t)$ appear to be correctly defined.

The source height, h , could be equal to ground height ($h = 0$) to delete the exponential term in equation 5.1. However, for most

distances, $h = 2$ m has little consequence and it can be left as such. The wind speed, $\bar{u}_1$, should be stability-class dependent (this is not clear in the text). Vertical dispersion, σ_z , is calculated incorrectly (see explanation in previous section). However, the empirical coefficients are correct and agree with other formulations.

6.5 Recommendations

- a. Keep the point-source sector-averaged Gaussian model to estimate air concentrations at distances >500 m from the tailings edge.
- b. If the receptor is <500 m from tailings edge, model the air concentration differently. This can be done in one of two ways:
 - i) use a parameterized numerical model, representative of the local conditions (e.g. Wilson 1982).
 - ii) use a multiple-line source Gaussian model (e.g. Van Ulden 1978).
- c. For the point-source model, calculate σ_z as a function of x , not $x + x_0$. Also, for all elements except radon, let $A_0 = \text{AEXPT}$. The code can be changed easily to accommodate these suggestions.
- d. For the radionuclides generated with dust (Ra, Th, U, Pb), only use the dispersion model with statistics representative of conditions when $\bar{u} > 5.4 \text{ m}\cdot\text{s}^{-1}$. A good approximation can be done by using wind classes ≥ 3 and stability class D.
- e. Ensure that $\bar{u}$ is stability dependent for Rn dispersion.

6.6 Summary

- a. Documentation and assumptions for the selected model require improvements.
- b. The model calculates air concentrations satisfactorily for distances >500 m from the tailings edge providing,
 - i) Radium, thorium, uranium and lead dispersion via dust is used only when $\bar{u} > 5.4 \text{ m}\cdot\text{s}^{-1}$ (i.e. when dust is generated).
 - ii) Radon dispersion follows time-averaged climatology
 - iii) σ_z is a function of x , not $x + x_0$.
 - iv) $\bar{u}_1$ is a function of stability
- c. The model does not work well for distances close to the tailings (<500 m). We suggest the use of either a parameterized numerical model, or a multiple line-source Gaussian model if the receptor is within 500 m of the tailings edge.

6.7 References

- Turner, D.B. 1970. Workbook of atmospheric dispersion estimates. U.S. Environmental Protection Agency. Office of Air Programs Publication No. AP-26.
- Van Ulden, A.P. 1978. Simple estimates for vertical diffusion from sources near the ground. Atmospheric Environment 12:2125-2129.
- Wilson, J.D. 1982. Turbulent diffusion in the atmospheric surface layer. Boundary-Layer Meteorol. 22:399-420.

7. Review of Soil Model (M.I. Sheppard)

7.1 Overview and description of requirements of component model

The soil model must provide estimated soil concentrations in a soil profile downwind of the tailings site. The source of the radionuclides is atmospheric deposition of radionuclide-enriched particulates only. Processes allowing removal of radionuclides from the soils are leaching, runoff and radioactive decay. The soil concentration is used to compute groundshine dose and vegetable ingestion dose.

7.2 Comparison of model with requirements

The soil model is a simple compartment model which neglects some important pathways and processes. It is not complete for this application. The model ignores soil contamination via contaminated regional groundwater or by contaminated irrigation water of gardens, both significant pathways for soil contamination. Sheppard and Dzik (1987) have shown that both of these mechanisms are important for radionuclide transport through soil. If groundwater is not contaminated at the receptor location, this condition should be stated explicitly (and proved preferably). We understand that agriculture is not included in the scenario, so that widespread irrigation of food crops need not be considered. However, small-scale irrigation of garden crops is a common cultural practice, and the water balance of soils in many parts of the Canadian Shield is such that irrigation is likely (Sheppard 1985). The model includes downward transport of nuclides through leaching, but ignores upward migration which clearly happens on a seasonal basis in humid areas, and continuously in arid environments.

The model assumes that there is no net flux of radon between atmosphere and soil; this is a reasonable assumption for an assessment model. For completeness, conservation of mass should be addressed briefly. Ignoring loss of radionuclides due to suspension is conservative. Lateral movement of radionuclides from soil to surface waters through erosion should be addressed.

7.3 Specific comments on model

The documentation does not discuss what impact the assumption of averaging the activity in the soil over time has on the model predictions. This could be substantial. The vegetation model gets the soil concentration from layer 2. This is not conservative; why not use a depth-weighted average? This is simply:

$$\frac{C_1 \cdot Z_1 + C_2 \cdot Z_2}{Z_1 + Z_2}$$

where C_1 and C_2 are soil concentrations in the layers 1 and 2, and Z_1 and Z_2 are the soil layer thicknesses. For high K_d nuclides such as uranium and thorium that will remain in layer 1 for a long time, the present method

will underestimate plant concentrations seriously. Layer 2 will always have substantially lower concentrations than layer 1.

A sensitivity analysis would have provided greater confidence in the suitability of the model, and shown the important driving parameters. The test of the model, Figure 6.3, shows odd results; why aren't thorium and uranium more similar? We assume it's because the source term for uranium is very much lower than for thorium. The model test is not described sufficiently to judge the success of the test because no input terms are given, nor is there any discussion or interpretation of the results.

7.4 Specific comments on parameters

Justification is not given for the parameter distributions chosen for the mineral soil physical properties. More rigorous definitions of the parameters are required (e.g. is soil bulk density wet or dry?). Also the need for parameter distributions is not given. Perhaps only one of the parameters, K_d , varies over more than an order of magnitude, and this is the only parameter needed to be treated in a probabilistic manner.

It is reasonable to set the deposition velocity to soil (V_g) equal to that selected for vegetation. We recommend that the distribution of V_g be changed to T (0.003, 0.02, 0.03) to coincide with that recommended for the terrestrial vegetation model (see section 9.4, this review).

7.5 Recommendations

- a. Incorporate irrigation water and groundwater contamination (e.g. Sheppard and Dzik, 1987).
- b. Compute a depth-weighted average soil concentration for plant uptake.
- c. Perform a sensitivity analysis of the model to determine its driving parameter(s). Discuss and interpret the test of the model.
- d. Rigorously assess the need for probabilistic treatment of each parameter.

7.6 Summary

- a. The model, as it is, is inadequate due to the omission of regional groundwater contamination and irrigation of gardens. Our recommendations would strengthen the approach taken.

7.7 References

- Sheppard, M.I. and E.J. Dzik. 1987. Soil contamination by irrigation and by capillary rise of groundwater. J. of Soil and Water Conservation, May-June issue.
- Sheppard, S.C. 1985. Use of FOOD III and SCEMR to assess irrigation as a biosphere pathway. Atomic Energy of Canada Limited Report, AECL-8380.

8. Review of Groundshine Model (J.W. Barnard)

8.1 Overview and description of requirements of the component model

The "groundshine" portion of the UTAP model calculates external dose resulting from proximity to contaminated soil. In this sense, groundshine is a misuse of term since the term skyshine has been traditionally reserved for reflection or backscatter off of the atmosphere. However, it has become common recently to use "groundshine" as the name for external dose-rate from contaminated ground arising from both scattered and unscattered gamma rays. We shall use the term in the same sense in the following comments.

A suitable groundshine model applicable to long-term dosimetric modeling should give reasonable estimates of the external dose rate to man from standing on top of contaminated ground. A reasonable estimate of dose rate would result from calculating dose rate at the surface from uniformly contaminated slabs or layers of soil of infinite lateral extent. The calculation method selected should be as accurate as possible, taking into account computational difficulties. Simplifying assumptions should tend to result in overestimation of dose rate.

Documentation of the model, if not the model itself, should account for all contributions to dose from all types of emissions even if the ultimate result is to ignore some components.

8.2 Comparison of the Model with Requirements

The groundshine model fulfills most of the above criteria. Its estimates, based on gamma dose rates from underlying slabs of contaminated soil, should result in reasonable estimates of dose rate at the surface. Values obtained should tend to be overestimates since attenuation in air and the differences in scattering properties between soil and air (Beck and de Planque 1985) have been ignored. We are unprepared, however, to say what effect replacing buildups, by the use of the linear absorption coefficients in place of the linear attenuation coefficients, will have on the safety margin of the results.

The documentation does not discuss the beta component of dose. Note that the dose rate in air at the interface of a semi-infinite medium uniformly contaminated with natural uranium and natural thorium is split approximately equally between the beta and gamma components. Massand et al. (1977) have estimated the dose rate at 5 cm above monazite sands in India to be 4.2 rem per year due to beta rays alone when the soil concentrations of thorium are about 17 Bq (they also found appreciable dose rates at a height of 1 m). Admittedly the principal dose is to the skin. When this is weighed into the calculation of effective dose equivalent with the weighting factor of 0.01, it becomes a small contributor. However, prolonged exposure effects should be discussed. Particularly, the effects of long-term exposures of exposed sensitive tissues such as the lens of the eye should be considered. Perhaps the question of beta doses was discussed in a previous phase of the study, but it should be also addressed here.

8.3 Specific comments on the model

8.3.1 General

A general comment is that the groundshine model is somewhat of a dinosaur. In making this comment we do not intend to reflect badly on SENES or the UTAP model in general. We are making the assertion realizing that we, ourselves, have been guilty of similar excesses in the recent past (Barnard 1986). In general, external dose modelers are reinventing the wheel with 1950s technology when 1970s technology is available for the same price. Price refers to the computing cost and the difficulties in constructing, documenting and understanding the models.

Previously, we noted that the model did not account for differences in the scattering properties of air and soil and touted it as a virtue because a slight overestimation in dose rate would result. However, calculations relating exposure rate in air at 1 meter above ground to contamination in soil have been calculated by Beck and de Planque (NCRP 1976; U.N. 1982). In fact, such dose-rate conversion factors have been calculated specifically for the uranium and thorium series. Once calculated, these factors should not require recalculation. Nevertheless, external dose modelers ignore these results, evidently, to the continuing chagrin of Beck and de Planque (1985).

8.3.2 Specific

Two specific comments of a more editorial nature are:

- (1) Page 7-5: The superscript referred to in paragraph 4 must occur in equation 7.4.
- (2) Page 7-10: The end of the first paragraph should likely be completed as "... and $1 y = 3.15 \times 10^7 s$ ".

8.4 Specific comments on parameters

We have reviewed all the parameters from the code listing (Table 7.2) and the accompanying correspondence that can be related back to existing tables of radiological data such as the Radiological Health Handbook (1970) or the Engineering compendium (IAEA 1968). This includes linear absorption coefficients and mass energy absorption coefficients. We have also checked conversion coefficients appearing in formulas. In general, all such parameters appear to have been assigned reasonable values. The mass energy absorption coefficient for flesh has been assigned a value of $3 \times 10^{-3} m^2 \cdot kg^{-1}$ for all energy groups. This is between 3 and 6 percent low over the whole energy range in comparison to the value given in the Radiological Health Handbook (1970) for muscle.

The values for specific activity for the four nuclide groups are correct. We couldn't check the average energy in all energy groups using the documentation we received because, except for group I, the group boundaries are not defined. Interestingly, the value calculated for N_{T1} in the example T1 on page 7-7 is in disagreement with the existing energy group calculations for thorium and daughters in the literature (NCRP 1976).

8.5 Recommendations

- a. The beta dose question must be addressed, if not within the code itself, at least within the documentation for the model.
- b. The values for the number of photons and average energies in each group should be checked to ensure they agree with values given in the literature (NCRP 1976). It would be helpful if the documentation listed the bounds of all the five energy groups used in the groundshine model.

8.6 Summary

- a. The model appears to be adequate to predict external gamma dose rate for long-term assessment of uranium tailings.
- b. A discussion of beta doses is required in the documentation.

8.7 References

- Beck, H.L. and de Planque, G. 1985. Dose Rate Conversion Factors, Health Physics Vol.49, pp 1015-1016.
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9. Review of Terrestrial Vegetation Model (S.C. Sheppard)

9.1 Overview and description of requirements of component model

The terrestrial vegetation model is required to predict the concentrations of radionuclide in plants resulting from deposition from the atmosphere onto plant surfaces, plus absorption from the soil through the roots. The concentration in plants is required to compute intake of radionuclides by man directly from vegetation, and via animal produce.

9.2 Comparison of model with requirements

The proposed model is the model used routinely in assessments over the last ten to fifteen years. The atmospheric pathway is time dependent, which is appropriate in this case, given the seasonality of wind erosion and plant growth. The root uptake pathway is assumed to be at a steady state. For annual plants, root uptake is seldom a steady-state process. However, the assumption of steady state is not inappropriate given the substantial variability of B_{vi} and the fact that most food crops are harvested at advanced ontogeny when radionuclide transfer is approaching a steady state.

The model is used to compute concentrations in food crops at various receptor locations throughout the assessment area, and to compute mean concentrations in animal-feed plants averaged over the assessment area. The model does not explicitly include cultural practices related to food production. Cultural practices include culture of native plants (for example: cranberry, blueberry, wild rice) and culture of garden plants. Some of these practices may not be done now, but are entirely feasible even in the most northerly areas. Once cultural practices such as gardening are considered, then irrigation must also be considered. Sprinkler irrigation using contaminated water will deposit radionuclides directly onto leaves. There is no doubt that garden plants will respond to additional water throughout Northern Ontario, despite the humid climate (Sheppard 1985).

9.3 Specific comments on model

The model is simple and is appropriate if the selection of parameter values is made with care. The root uptake pathway is based on the transfer factor B_{vi} . The studies reported in the literature that measure B_{vi} differ markedly in the scenarios they emulate. A key difference is the distribution of the radionuclide in the soil. The most common way to compute B_{vi} is to base it on a mean soil concentration in the 0 to 15-cm layer. This is because agricultural soil tillage homogenizes this layer. This layer may not be well mixed in tailings scenarios where the radionuclide may be i) present only in the litter layer as a result of atmospheric deposition and no soil mixing, or ii) present in a subsoil layer either on the tailings or due to groundwater contamination downslope from the tailings. Plants can derive radionuclides from either litter layers or subsoil layers, and this may require more attention to the modelling of migration in soil. For example, the present model appears to exclude the litter layer from the soil rooting zone. This is non-

conservative for atmospheric deposition cases and probably not appropriate. To counterbalance this criticism, the high degree of variability in B_{vi} values probably encompasses some of the effects due to non-uniform distributions of roots and radionuclides in soil.

A potentially-potent pathway that seems to have been ignored is the transfer of radionuclides from the aquatic pathways to terrestrial plants. Since the aquatic pathway, by way of streams, seems to be an important vector in the tailings scenarios, the linkage between aquatic and terrestrial pathways should be considered. The simplest transfer scenario to consider is seasonal flooding that will carry contaminated sediment onto otherwise terrestrial settings. Changes in stream course can also occur. An example of this kind of transfer was documented near Oak Ridge, Tennessee where sediments contaminated with mercury less than thirty years ago are now found as terrestrial soils (G. Blaylock, 1986; personal communication). The transfer occurred because the stream changed course slightly, there was seasonal flooding, and the former sediments were mined for use in gardens. A more dramatic example is the disfiguring itai-itai disease (Cd toxicity) experienced by people living downstream of zinc mines and using the stream waters and sediments on food-production land (Kitagishi and Yamane, 1981). Other transfers from the aquatic pathways to terrestrial plants include root intrusion on the stream banks and irrigation.

9.4 Specific comments on parameters.

The proposed values for the parameters (symbols are those of SENES 1986 documentation) F_{ry} , E_v , t_v , V_d , Y_v , λ_{vv} , and B_{vi} were reviewed. The values presented were selected to model uptake by vegetables, forage, browse and lichens. The proposed values generally were appropriate.

The value of t_v for forage and browse (t_a and t_b , respectively) have been set at 30 d (2.592×10^6 s). This is the usual value for pasture and hay crops that are utilized on about that interval throughout the summer. This may not be appropriate for game feeding, where game may utilize the plants only once each year. Longer times are conservative and we recommend 100 to 200 d, the snow-free interval, which is more appropriate in this scenario.

The values of B_{vi} are based on a literature compilation by Cassaday et al., (1985). The values are reasonable. However, no apparent effort was made to screen, qualify or appropriately weight the data derived from various literature sources by Cassaday et al. (1985). For example, means from other compendiums were averaged on an equal basis with observations made in individual studies. The literature search by Cassaday et al. (1985) was not exhaustive. Further work should be done to derive the best estimates available for B_{vi} .

The deposition velocity values (V_d) were based on the literature, and the range selected is reasonable. The reference used as a basis for dry deposition (Sehmel 1980) also suggests a mechanistic model to calculate V_d . The results of such a model, using appropriate meteorological parameters, actually gives a V_d distribution close to a log-normal shape. The UTAP model uses a uniform distribution of V_d , resulting in

insufficient probability of occurrence of values near $0.02 \text{ m}\cdot\text{s}^{-1}$. It is suggested that a triangular distribution would be more satisfactory using the values given so that the V_d distribution would be $T(0.003, 0.02, 0.03)$.

On page 8-4, the authors misrepresent the results of Tracy *et al.* (1983) and Sheppard and Thibault (1983). Neither paper examined the statistical distribution of radionuclide concentrations in plants. A similar misrepresentation of these papers was given by Cassaday *et al.* (1985).

9.5 Recommendations

- a. Clearly justify why plant cultural practices are not considered, since this omission is not conservative.
- b. The litter layer must be included when computing the concentration of radionuclides in the plant root zone.
- c. Justify that plant uptake from contaminated subsoil (ie. below 15 cm) is either an unlikely pathway or model it in future revisions.
- d. Clearly justify why the transfer from aquatic pathways to terrestrial plants, especially by way of sediment, is not considered since this omission is likely to be not conservative.
- e. Continually update values of $B_{v,i}$ as new data become available.
- f. Revise the values of t_a and t_b .
- g. Change distribution of V_d to $T(0.003, 0.02, 0.03)$.

9.6 Summary

- a. The model for terrestrial vegetation is appropriate but some attention must be given to the modelling of radionuclide distributions and concentration in the root zone.
- b. The transfer of radionuclides from aquatic sediments to terrestrial plants has not been modelled, and this linkage is potentially important and realistic.
- c. Cultural practices that include irrigation have not been considered.
- d. Changes to some parameter values are recommended.

9.7 References

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- Tracy, B.L., Pranti, F.A., and Quinn, J.M. 1983. "Transfer of Ra-226, Pb-210 and Uranium from Soil and Garden Produce; Assessment of Risk." Health Physics, Vol. 44, pp. 469-477.

10. Review of Subsurface Transport Model (T.W. Melnyk)

10.1 Overview and requirements.

The subsurface transport model calculates the flow rates, or concentrations, of all contaminants at the discharge point of the groundwater flow system. The model must take into account the migration behaviour of radionuclide decay chains including the ingrowth of daughter nuclides.

Source concentrations and the volumetric seepage rate are provided by the source model. The outputs from the subsurface transport model are used as inputs to the river/lake surface water system of the surface water dispersion model. Concentrations are converted to fluxes for the surface water dispersion model.

10.2 Comparison of model with requirements

The subsurface transport model is an assessment model, rather than a research model, and is hence a simplified representation of the actual system. A single groundwater flow path, through a glaciofluvial sandy aquifer, is used with constant groundwater flow rate and dispersion coefficient. Constant retardation factors are used. Concentrations are calculated at points on the flow path as a function of time. The point chosen for calculations is the discharge point of the groundwater flow path. The discharge point is selected to lie between the toe of the tailings dam and a river 200 m away so the groundwater transport path is relatively short. All seepage from the tailings pile is assumed to form the groundwater passing through the "window" under the dam. The model does not account for dilution with other sources of groundwater nor any lateral dispersion.

The general concept and implementation of the model are acceptable in scope for an assessment model. However, there is inadequate consideration of the migration of radionuclide chains and ingrowth of daughter nuclides. The documentation presented for peer review also contained a number of inadequacies and inconsistencies. Specific items are addressed in the following sections.

10.3 Specific comments on model

10.3.1 Transport model

In Section 1.0 Introduction on page 1-2 (SENES 1986 report), it is claimed that the models can deal with input parameters that vary with time. This statement is not correct for the subsurface transport model using the Green's function approach described. The Green's functions are derived for a zero initial condition with constant parameters.

Equation 9.2 gives the Darcy velocity of the groundwater through the "window" under the dam. If the cross-sectional area of the plume changes, this velocity will change. Figure 9.2 shows this cross-section

changing. However, the Darcy velocity has been considered a constant in solving the mass-balance equations.

The flux, J , in equation 9.11 is defined with respect to the cross-section of the "window" below the dam. It does not give "flux" at the discharge area. The total integrated flow rate in units such as $Bq \cdot a^{-1}$ would be a better variable to use.

Page 9-3 includes the statement that chemical or sorption partitioning of elements between groundwater and immobile phases is "for the most part" fast, reversible, and exhibits a linear isotherm. This statement is inconsistent with the previous discussion of the contribution of carbonate dissolution and coprecipitation. Neither of these phenomena is described by an isotherm in the usual sense, and certainly not a linear one. In addition, neither mechanism is generally reversible unless the general groundwater chemistry changes. There is insufficient information provided to assess the importance of these phenomena to the transport. Either these mechanisms are important and should be addressed in the transport model or they are not important and the discussion should reflect this. The inclusion of any mechanisms other than "linear sorption" would cause considerable mathematical difficulty in the transport model.

The heterogeneity of the medium is what creates and defines the flow paths. Consequently, the effects of the inherent heterogeneity cannot be taken into account by use of stochastic parameters if the medium is considered homogeneous and isotropic for each choice. The groundwater flow paths will take the paths of lowest permeabilities so water velocities and permeabilities used should generally be biased towards the more permeable end of the distribution observed in the field. The statement on page 9-9, that effects will be partially accounted for, is reasonable but more discussion of the implications of this assumption, addressing the points raised here, should be given.

The parameter X is the distance between the toe of the tailings dam and the groundwater discharge point. The average groundwater flow path length begins under the tailings pile, passes under the dam, and resurfaces at the discharge point. Consequently, the travel path would be somewhat longer than distance X . However, X has been used as the migration distance in the Green's function evaluation in the implementation of the model. It would be better to calculate the real migration path length X and use this longer migration distance in the Green's function. However, X is the shortest possible path length and use of X as the migration distance will produce a conservative result.

The distribution of migration starting points under the tailings pile, all with different migration path lengths to the discharge point, would add a significant dispersion to the results calculated at the discharge point. This additional dispersion has not been accounted for or discussed. The additional dispersion will not affect the maximum concentration reached at the discharge point but will produce earlier arrival of the first contaminant. However, since the shortest migration path length is used for the transport calculation (see previous comment) all the contaminant discharges at the earliest possible time and the result is conservative.

The detailed solution to the mass balance transport equation, given in equation 9.12 was not checked, since it was adequately referenced.

In equation 9.13, $G(X,t)$ is not a Green's function in its normal definition. A Green's function is the solution to the mass balance equation with specified boundary and initial conditions for a delta-function or instantaneous source (Goertzel and Tralli, 1960). It appears in the general solution as an integrand together with the continuous source function in the following form:

$$O(t_k) = \int_0^{t_k} I(t') g(t_k - t') dt' \quad (1)$$

where $O(t_k)$ is the outlet concentration at time t_k , $I(t)$ is the continuous inlet concentration, and $g(t)$ is the Green's function. The dependence of $O(t)$ and $g(t)$ on X has been dropped from the notation for simplicity. Integrating this equation by parts where

$$u(t') = I(t') \quad \text{and} \quad dv = g(t_k - t') dt' \quad (2)$$

gives

$$O(t_k) = - I(t') G(t_k - t') \Big|_{t'=0}^{t_k} + \int_0^{t_k} (dI/dt') G(t_k - t') dt' \quad (3)$$

where the integrated Green's function, $G(t_k - t')$ is defined by

$$G(t_k - t') = -v = \int_0^{t_k - t'} g(t'') dt'' \quad (4)$$

In the application described, the integration is approximated by considering $I(t)$ to be approximated by a constant, so that $dI/dt = 0$, except at discrete values of time, t_i , where $dI(t_i)$ is ΔC_i . Consequently, the integral can be replaced by a discrete sum over index i . Also, since $G(0) = 0$, Equation (3) becomes

$$O(t_k) = I(0)G(t_k) + \sum_i \Delta C_i G(t_k - t_i) \quad (5)$$

Equation 5 above corresponds to equation 9.15 of the Senes report, where the Green's function used in equation 9.15 is the integrated Green's function defined by equation 4 above. It is not clear why the last term in equation 9.15 has been divided by a factor of two. The comment in the last sentence of Section 9.4.2 is inadequate in addressing this point.

The continuous source is illustrated as being monotonically increasing. Equation 9.15 is robust for this case but will give a bias to lower values of the output concentrations. It would be better to approximate the integral in equation 3 (above) by a better scheme, such as a trapezoidal rule. If the source goes through a maximum concentration and then begins to decrease, then subtraction of terms will occur in equation

9.15 and the results could become unstable for some times. A more robust form of the equation, directly approximating equation 1 (above), would be better.

The results shown in Figure 9.5b of the SENES report are not consistent with the data in Table 9.5. For example, from the data in Table 9.5, the sorption constant, K_d , for uranium is $0.02 \text{ m}^3 \cdot \text{kg}^{-1}$, the porosity, n , of the aquifer is 0.33 and the soil density, ρ_s , is $2650 \text{ kg} \cdot \text{m}^{-3}$. Thus, from equation 9.5 and the relation $\rho_b = (1 - n)\rho_s$, the retardation factor, R , for uranium is 109. The volumetric seepage rate, from equation 9.3 is $3 \times 10^4 \text{ m}^3 \cdot \text{a}^{-1}$ and the Darcy velocity of the groundwater, from equation 9.2 is $30 \text{ m} \cdot \text{a}^{-1}$. Hence, the groundwater velocity, q/n , used in the transport equations 9.9, 9.10 and 9.12 is $90.9 \text{ m} \cdot \text{a}^{-1}$. With the migration path length, X , of 80 m, the water transit time is $80/90.9 = 0.88$ years. Hence, the predicted breakthrough time for uranium is at $(0.88)(109) = 96$ years. Figure 9.5b shows the 50% breakthrough time at about 210 years.

Figure 10.1 (this report) illustrates the results obtained with SYVAC3-CC3 using the data given in Table 9.5. In this figure, the 50% breakthrough of uranium occurs at about 85 years consistent with a slight shift to earlier breakthrough time due to the effects of dispersion. Similarly, the breakthrough time for Ra-226 is predicted to be at 440 years using the data in Table 9.5 but Figure 9.5b shows 50% breakthrough occurring at about 950 years. Figure 10.1 (this report) shows the SYVAC3-CC3 calculated 50% breakthrough time at about 320 years, consistent with the prediction. A similar inconsistency exists for the other two nuclides, with the SYVAC3-CC3 calculation showing thorium beginning to breakthrough within the 1000 year simulation time. If Figure 9.5b is not the wrong figure to correspond to the data in Table 9.5, then there are errors with the evaluation and implementation of the subsurface transport model. The inconsistency must be resolved or else the quality assurance of the subsurface transport model in the UTAP code is compromised.

10.3.2 Radionuclide Chains

Section 2.0 of the SENES documentation states that the model deals with radionuclides uranium, thorium, Ra-226, Rn-222, Pb-210, and the aqueous species of calcium, iron and sulphate used to calculate pH in the surface water bodies. The radionuclides are not handled as a decay chain. Thus, the complete determination of the ingrowth and migration of daughters is not possible. Further discussion of the problems caused by this lack of chain migration is presented below. There were no data relating to the migration of calcium, iron or sulphate and no description of how or if the subsurface migration of these species from the tailings pile was included.

There was insufficient explanation of the ingrowth of Pb-210. The origin of equation 9.7 was not explained adequately. We were able to derive it by assuming that the decay constant for Ra-226 is negligible with respect to the decay constant for Pb-210. The tables give the ratio as 0.014 so the approximation is reasonable. This extra assumption underlying equation 9.7 should be stated.

Equation 9.8 gives an approximation to the ingrowth of Pb-210. However, this Pb is strongly retarded. The Pb produced near the discharge

point will lag behind the parent Ra-226. Thus, Equation (9.8) will produce a breakthrough of Pb-210 that is too early. Figure 10.2 (this report) illustrates the results of calculations performed with SYVAC3-CC3 by using equation 9.8 and by using the Ra-226/Pb-210 decay chain for the parameter set given in Table 9.5. The results of equation 9.8 are a reasonable approximation for this case, but the results are a strong function of decay constants and retardation factors.

In the code listing in the SENES report, it appears that equation 9.8 is coded with a "DT" in the exponential argument while the equation in the text has "T". This may be correct in the manner that the equation is implemented but should be checked. A better explanation of the implementation of this ingrowth term in the text is needed.

There is a significant contribution to Ra-226 concentrations due to ingrowth from Th-230. The statement on page 9-7 discounting this ingrowth is wrong. The source model description indicates there is significant activity from Th-230. It is considered to be in secular equilibrium with the Ra-226. There is then a further contribution to Pb-210 from ingrowth from this ingrown Ra-226. Figures 10.3 to 10.5 (this report) obtained with SYVAC3-CC3, illustrate the breakthrough and the relative contributions to total Ra-226 and Pb-210 from the various sources using a $10 \text{ Bq}\cdot\text{m}^{-3}$ continuous source for all four nuclides, including Th-230. Other parameters are as given in Table 9.5. The activity and transport of Th-230 should be included separately from the Th-232 transport.

10.3.3 Documentation

Page 9-3 of the SENES report should state distribution coefficients are constants and give some references to use of this approach to back up the phrase "widely used". References should be given for introduction of retardation factors in this form into equation 9.10 if a better explanation is not given. In Table 9.1, the assumption that the Darcy velocity is considered to be constant should be stated.

Definitions of activities, A, following equation 9.7 do not state whether the volume used is porewater volume or bulk volume. The explanation preceding equations 9.8a and 9.8b is unclear in explaining how these equations were derived. Perhaps "dividing" by soil volume should be "multiplying" by soil volume? What is the difference between symbols A and C in equations 9.7 and 9.8a? They are both defined as activities and have the same units of $\text{Bq}\cdot\text{m}^{-3}$. Symbol C in equation 9.9 is defined as activity, but in equation 9.11, C is defined as concentration also with units of $\text{Bq}\cdot\text{m}^{-3}$.

In the second paragraph on page 9-9, "heterogenicity" should be "heterogeneity". Multiple discharge points would require a change in only the implementation of the model since it is already stated that the model is appropriate for the other discharge points by use of parallel runs. Porosity and hydraulic gradient are dimensionless quantities and should have no units. The units given (e.g. $\text{m}^3\cdot\text{m}^{-3}$) are inappropriate.

On page 9-21, what does "quasi-constant" mean for the dispersivity? It appears to be a time-independent calculated parameter.

In the middle of page 9-23, the word "contributed" should be "attributed". The distribution for parameter f has standard deviation given as 1.27 in Table 9.2 but the text on page 9-16 indicates the standard deviation is 1.66.

10.4 Specific comments on parameters

The cross-sectional area of the flow path "window" beneath the dam should be a distributed parameter since it is clear this is not a well-known quantity. The generic aspect of the parameter distribution for this quantity would add additional variability.

The half lives for U-238 and Pb-210 given in Table 9.4 do not agree with those given in the "Chart of the Nuclides" published by General Electric (Walker et al. 1977). This publication gives 22.3 years as the half life of Pb-210 (rather than 21 years) and 4.468 billion years for the half life of U-238 (rather than 4.51 billion years). These small differences in half lives would affect the results only very slightly.

The density used in equation 9.5 is termed bulk density. However the data tables and parameter descriptions give soil density with a different symbol. Presumably these two densities are related by a $(1 - n)$ factor, where n is porosity, but no equation is given in this section. Table 9.5 uses a different symbol again but the quantity is soil density; the table likely has a typographical error.

The definition of soil density on page 9-16 does not make it clear whether it is the density of bulk soil or the average density of the soil particles themselves. The context in other places indicates that it is the latter. A phrase such as "the ratio of the mass of the soil particles to the volume occupied by only the soil particles themselves" would be clearer. In the code the soil density is variable SDGWAT, but Table 9.2 indicates it is variable SGGWAT.

10.5 Recommendations

- a. The inconsistency in the demonstration calculation between Table 9.5 and Figure 9.5b must be cleared up.
- b. The Th-232 chain and Th-230 in the U-238 chain, should both be included specifically in the subsurface transport model. Consideration of the ingrowth of radium and of lead from throrium might be approximated by ingrowth equations analogous to equation 9.8 (Giuffre and Ross 1979). However the transport calculation for chains is done more exactly using a Green's function for chain members. These functions are well known (see Burkholder and Rosinger 1980, and Heinrich and Andres 1985, in addition to the references quoted in the SENES report) and would not add an insurmountable complexity to the code.
- c. The cross-sectional area of the "window" below the dam should be a distributed parameter or should be calculated from distributed parameters

10.6 Summary

- a. The general concept and implementation of the subsurface transport model are acceptable in scope for an assessment model. A number of relatively minor points and suggestions for improvements were described in the preceding sections.
- b. However, there is inadequate consideration of the migration of radionuclide chains and ingrowth of daughter nuclides, in particular from Th-230 in the U-238 decay chain.
- c. The documentation presented for peer review contained a number of inadequacies and inconsistencies. A major inconsistency exists in the results of the demonstration calculation.

10.7 References

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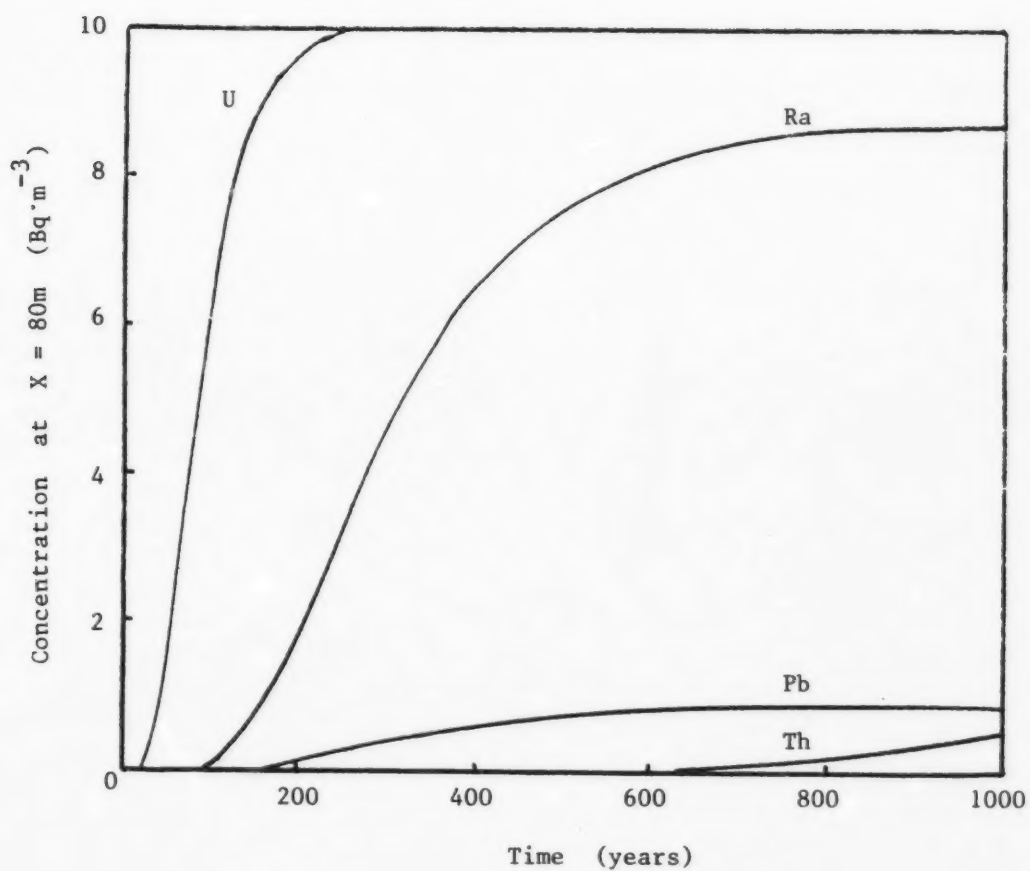


Figure 10.1. Concentrations of U-238, Th-232, Ra-226, and Pb-210 at a groundwater discharge point 80 m from the inlet. Data were taken for the UTAP deterministic demonstration case from Table 9.5. The radium $\rightarrow$ lead decay chain was used. Each nuclide had a constant concentration of $10 \text{ Bq}\cdot\text{m}^{-3}$ at the inlet.

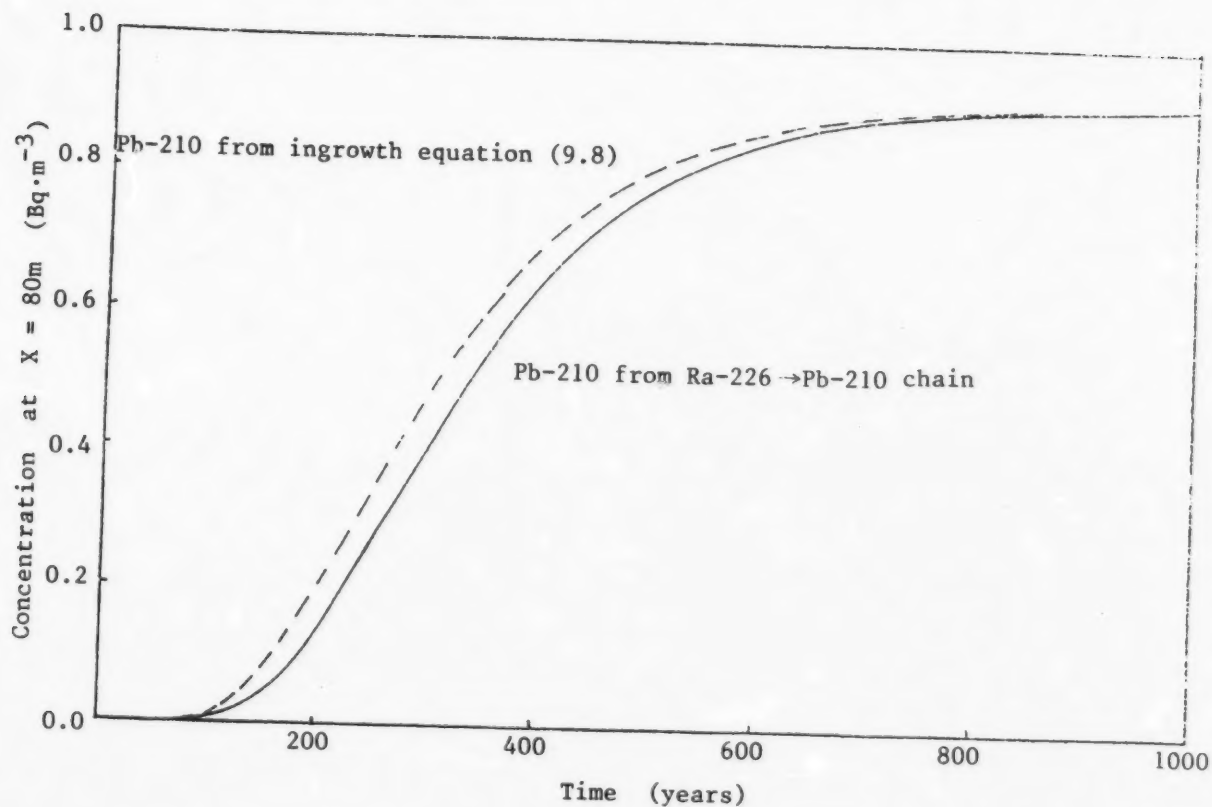


Figure 10.2. Concentrations of Pb-210 at 80 m from the inlet obtained from the Ra-226 concentrations by using equation 9.8 and from the Ra-226 → Pb-210 chain calculation.

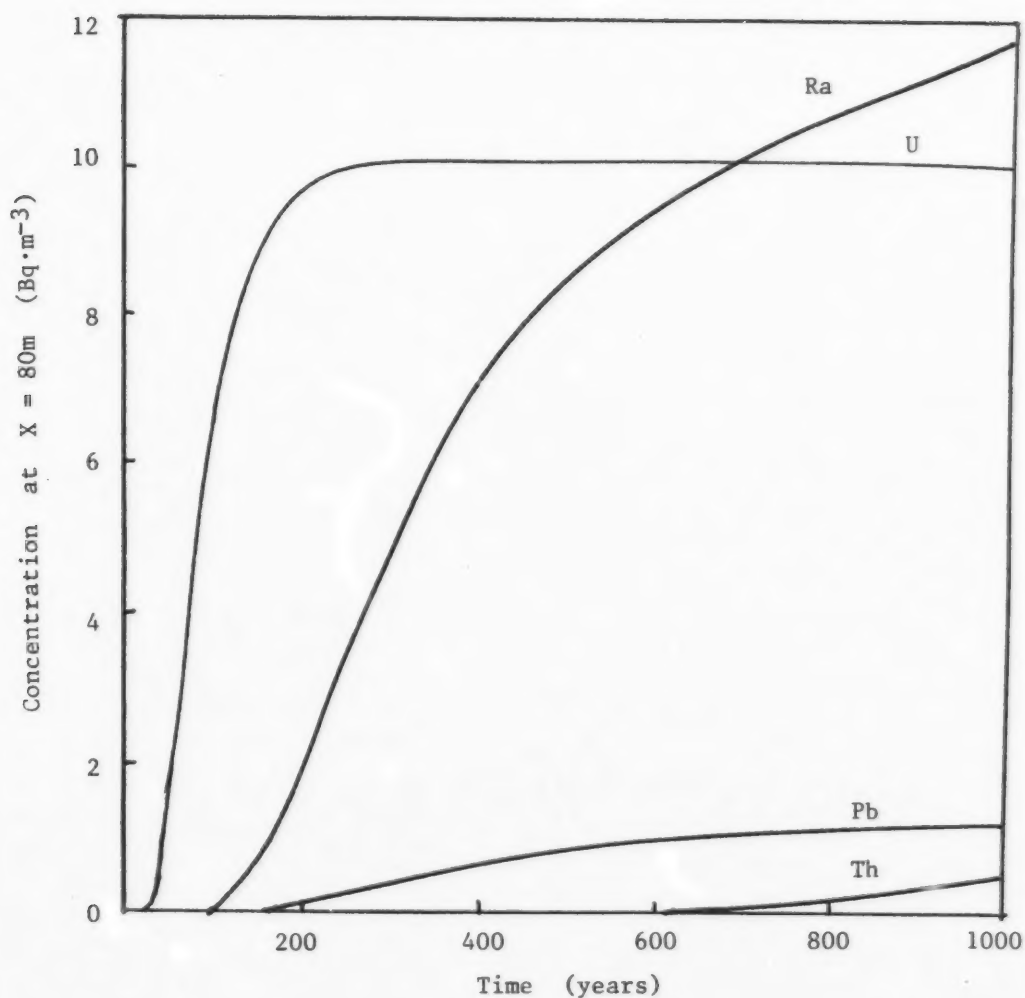


Figure 10.3. Concentrations of U-238, Th-230, Ra-226, and Pb-210 at a groundwater discharge point 80 m from the inlet. Data were taken for the UTAP deterministic demonstration case from Table 9.5. The uranium $\rightarrow$ thorium $\rightarrow$ radium $\rightarrow$ lead $\rightarrow$ four-member decay chain was used. Each nuclide had a constant concentration of $10 \text{ Bq}\cdot\text{m}^{-3}$ at the inlet.

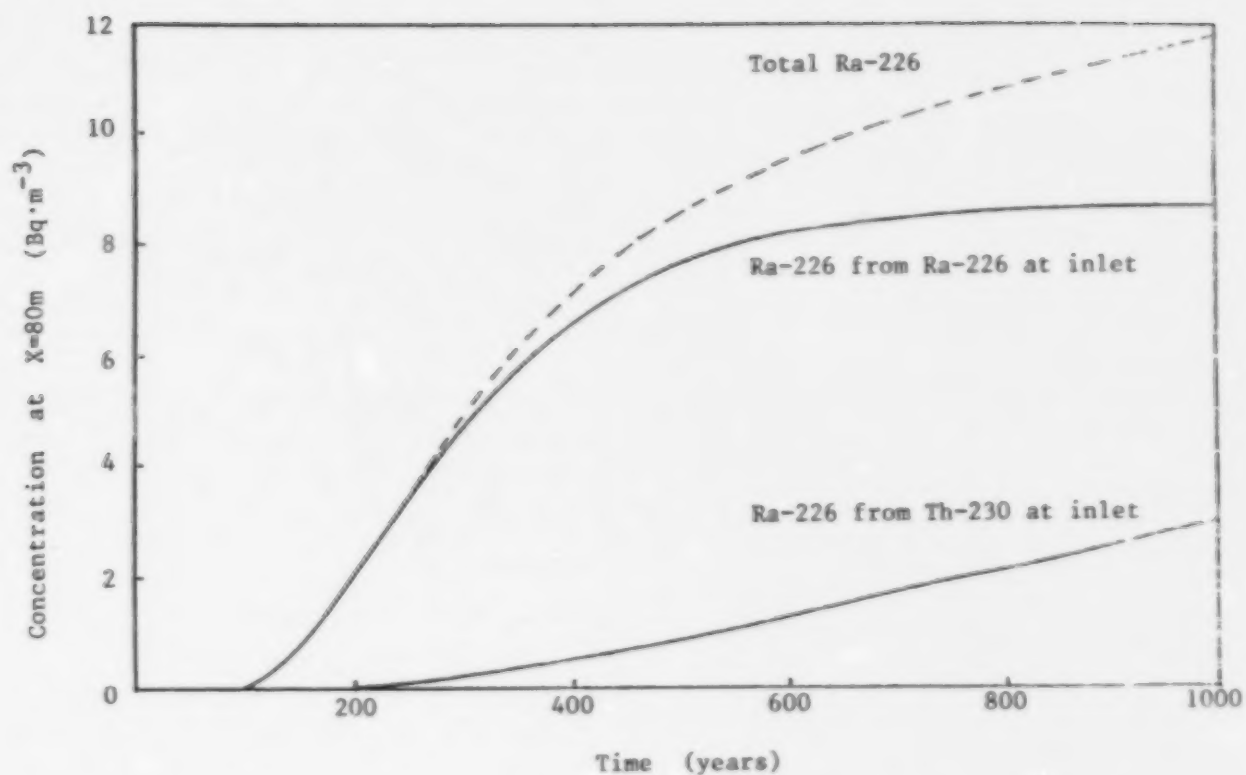


Figure 10.4. Concentrations of Ra-226, at 80 m from the inlet showing the contributions to the total Ra-226 from the Ra-226 at the inlet and from ingrowth from the Th-230 at the inlet.

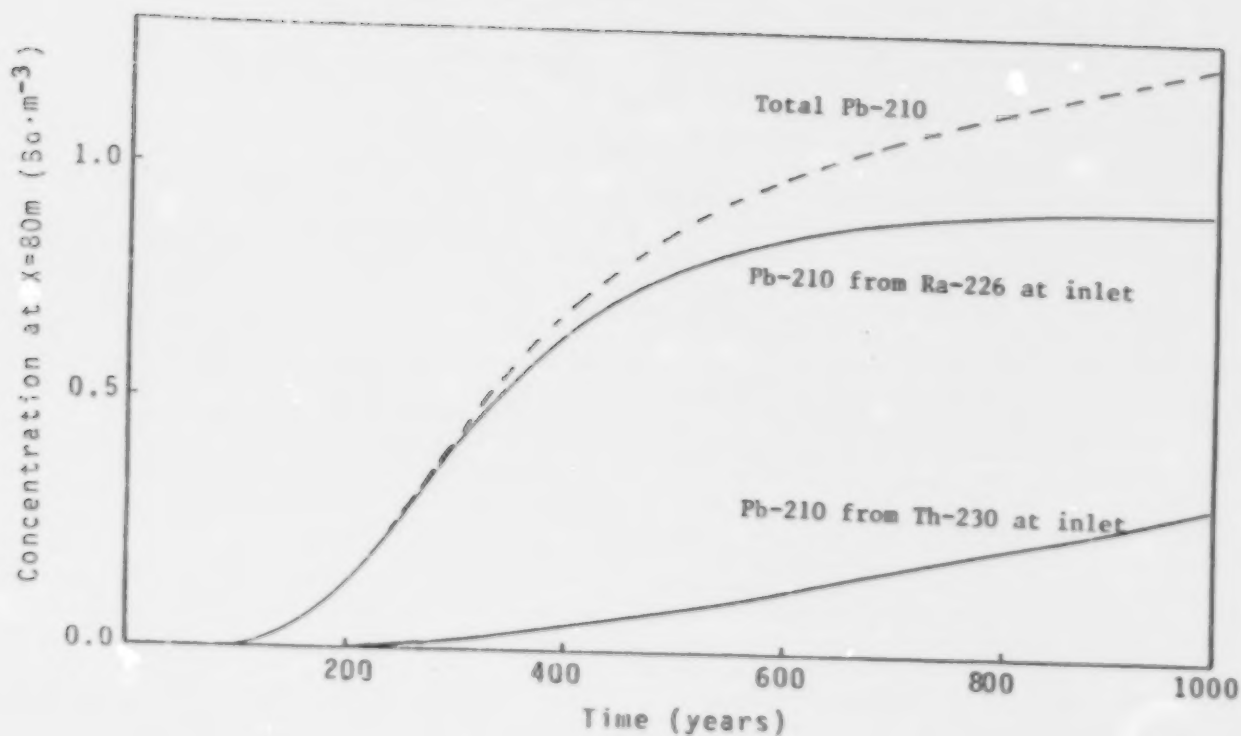


Figure 10.5. Concentrations of Pb-210 at 80 m from the inlet showing the contributions to the total Pb-210 from ingrowth from the Ra-226 at the inlet and from ingrowth from the Th-230 at the inlet. The Pb-210 at the inlet is strongly retarded and does not breakthrough.

11. Review of Surface Water Dispersion Models (R.J. Cornett)

11.1 Overview and description of requirements of component model

The surface water dispersion model is required to estimate the concentrations of radionuclides (uranium, thorium, Ra-226 and Pb-210) and pH in rivers and lakes located downstream from a tailings area in a reference drainage basin located on the Canadian Shield. These models are appropriate for a string of interconnected rivers and lakes located sufficient distance from the tailings source that the longitudinal concentration gradients between nodes and transverse gradients in a lake or river are negligible. The outputs from the models are the concentrations of nuclide in the water at each node. This output is used as input for modelling the fish, aquatic plants and drinking water ingestion pathways.

11.2 Comparison of model with requirements

The models developed are an appropriate description of the long-term dynamics of nuclides and ions in surface waters. The accuracy of predictions may be poor at the first node if: a) the distance between the tailings and the first node is a short stretch of a river as illustrated in Figure 10.1, or b) the residence time of water in the first reach is greater than the time-step. Since this river model is formulated to ignore transit times, this assumption could be very important if radionuclide discharges vary significantly over a timescale shorter than the model time-step. We suspect that the accuracy of the predictions at node 1 will be controlled by the inhomogeneities in this reach as the nuclides will not be uniformly dispersed in the system. The pH values calculated will be conservative (i.e. pH actual is greater than pH calculated) for the reasons described below.

One important assumption, inherent in this model, is that the physical setting/morphometry of the lake and its basin are static. Although this is correct over hundreds of years, it is not correct over longer time periods. Lakes may fill with sediment and be used as agriculturally productive areas. This could enhance radionuclide transfer to man.

11.3 Specific comments on the model

These models are based upon mass balance principles and employ:

- a) a single continuously mixed dilution reservoir for river stretches, and
- b) a two compartment (water and active sediments) continuously mixed reservoir model for lakes.

Although the assumptions inherent in the models are poorly stated, these assumptions necessary to formulate this assessment model are inherent in most long-term assessment models. These assumptions are also supported by the long-term simulations of nuclides in several studies (Cornett and Ophel 1986; Edgington 1981, Valhgren et al 1980, Carlsson 1978).

The hydrologic input to the lakes assumes that annual net precipitation onto the lake surface is negligible. Usually P_t (lake) $\approx$ 0.1 m/year in the Ontario region of the Canadian Shield. The significance of this assumption is dependent upon the ratio of lake: watershed area. If this ratio is large, the volumetric discharge will be underestimated and nuclide concentrations in the lake water will be overestimated. In the characteristic watershed this bias is small since the ratio of Area lake: Area watershed = 40:1.

The model to predict surface water pH assumes iron and aluminum in equilibrium with goethite and gibbsite. Concentrations of sodium, potassium, magnesium and chlorine are essentially unaffected by the tailings area drainage and are characterized by the background levels.

Concentrations of dissolved calcium and sulphate depend on gypsum dissolution and pyrite oxidation. The calculation of Ca^{2+} and SO_4^{2-} concentrations assumes that these substances are not removed from the water by any biogeochemical processes in lakes or rivers (i.e. the influent and effluent concentrations are equal). This assumption is not correct. In a surface water system, concentrations of Ca^{2+} and SO_4^{2-} will increase 10-20% along a given watershed (Turkey Lakes watershed, D. Jeffries, pers. comm.). Calcium exchanges with other ions on the sediments and SO_4^{2-} reduction in the sediments is significant in most lakes (Rudd et al. 1986). Co-precipitation with newly formed minerals will occur with increasing pH (IEC Beak 1984). The contribution of groundwater to the system (especially in small streams) should not be neglected. The pH estimates based on the Ca^{2+} and SO_4^{2-} ions will be biased. The models will substantially underestimate actual pH's in lakes (perhaps by more than 1 pH unit) receiving Ca^{2+} and SO_4^{2-} inputs from the tailings discharge.

Factors which could also contribute to pH changes (short-term events) would be the accumulation of a low pH snowpack (Jeffries et al. 1979) and a large input of acidic precipitation (Scheider et al. 1979). A change in H^+ and SO_4^{2-} concentrations in precipitation will alter the inputs of these substances and other ions from the background catchment regions. The model assumption, that these inputs will remain constant, is unlikely.

In the radionuclide analysis it is difficult to justify excluding Po-210 from the surface water analysis since Po-210 is rarely in secular equilibrium with its Pb-210 parent in surface waters (Talbot and Andren 1984). Most of the important dose consequences result from exposure to the daughters and granddaughters of Po-210. Therefore exclusion of this nuclide or assumptions of secular equilibrium with Pb-210 are questionable. The significance of this assumption needs to be tested using a sensitivity analysis.

The nuclide mass balance analysis assumes the active layer of the sediments is a continuously mixed box. Therefore it is inconsistent to include a different term $[(R_i)_{\lambda}, t]$ to describe the nuclide concentration at the bottom of the active sediment layer than that used at the top of the sediment layer $[(R_i)_1, t]$.

The net primary production estimate $40 - 200 \text{ g} \cdot \text{m}^{-2} \cdot \text{a}^{-1}$ is too high in shield lakes (e.g. Fee 1980). As a result, the estimate of σ_s and

σ_p is about two times too high. A log-normal distribution is also more appropriate than a triangular distribution (Cornett, unpublished data). The parameterization of the diffusive flux of radionuclides from the sediments to the water using a surface renewal time has not been rigorously tested. An alternate more defensible approach developed by Santschi et al. (1983) uses diffusion through the benthic boundary layer. There are more data to support this approach (Santschi et al. 1986).

The value of u , the mean current velocity, is 10 to 100 times too high for small lakes. As a result, the back diffusion of nuclides from the sediments to the water will be overestimated by a proportional amount. This will overestimate dose consequences to the receptors.

The term $[R_i]_1$ is used initially to specify the sediment pore water concentrations in equations 10.1 to 10.18 and then it is used to represent the total activity (i.e. interstitial plus solid matrix activity) on the sediments. The redefinition is clear if one derives the equations. However an explicit statement in the documentation would be useful and should be made for equation 10.21.

The ingrowth terms described in the "SURFACE WATER DISPERSION MODEL" are not consistent with those used in the SOURCE module, and should read:

$$\begin{array}{ccc} \text{.....} & \frac{\lambda_{Ra-226}}{\lambda_{Pb-210} - \lambda_{Ra-226}} & \text{.....instead of} \quad \frac{\lambda_{Pb-210}}{\lambda_{Pb-210} - \lambda_{Ra-226}} \end{array}$$

Similarly for the ingrowth of Ra-226 from Th-230 the equations on the bottom of page 10-30 should read:

$$\begin{array}{ccc} \text{.....} & \frac{\lambda_{Th-230}}{\lambda_{Ra-226} - \lambda_{Th-230}} & \text{.....instead of} \quad \frac{\lambda_{Ra-226}}{\lambda_{Ra-226} - \lambda_{Th-230}} \end{array}$$

11.4 Specific comments on parameters

Site specific variables were not reviewed. The radionuclide and element concentrations calculated in the equations are dimensionally and mathematically correct.

P_t , mean annual net precipitation, is slightly higher than values reported in other studies (Beals 1984). In our opinion, the current velocities (u) are 10 to 100 times higher than those found in the deeper areas of lakes. Since most nuclides will be eventually deposited in these areas, rather than in the epilimnetic or littoral zones, we recommend lower values of u ranging from 3.2×10^2 to 3.2×10^4 . The thickness of the active sediment exchange zone (λ) could be specified as a variable rather than a constant. Published estimates of λ range from 0 to 10 cm (e.g. Robbins 1980). As noted earlier, estimates of σ_s and σ_p are ≈ 2 times higher than median sediment accumulation rates calculated from Pb-210 dating.

11.5 Recommendations

- a. The bias inherent in Ca^{2+} and SO_4^{2-} calculations should be tested.
- b. The significance of omission of Po-210 from the surface water model should be tested using a sensitivity analysis.
- c. The value $[\text{R}_i]_\lambda$ should be omitted and replaced with $[\text{R}_i]_1$.
- d. Parameters describing the accumulation of sediment should be revised.
- e. The rates of back diffusion calculated using the surface renewal time model should be checked with those estimated by Santschi et al. (1983).
- f. The incorrect equations describing decay should be changed in the computer code.

11.6 Summary

- a. Although documentation of the model and inherent assumptions is weak, this model can be expected to predict reasonably-accurate radionuclide concentrations in surface waters located several km below a tailings area, provided that the lake does not fill with sediment so that the physical characteristics of the lake system change. We expect some change over thousands of years, and extrapolation of the UTAP models over these time periods should not be attempted.
- b. The parameters/variables u , σ_D , σ_s , $[\text{R}_i]_\lambda$ and $[\text{R}_i]_1$ should be revised.
- c. The critical issue missing from this analysis is any testing/verification of this model against measured radionuclide concentrations in freshwaters. A comparison against Ra-226 in the Serpent River system is strongly recommended.

11.7 References

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12. Review of Aquatic Models (S. Mihok and R. Zach)

12.1 Overview and description of requirements of component model

The aquatic model is required to estimate radionuclide concentrations in game fish flesh and aquatic macrophytes through the use of bioaccumulation factors relative to water concentrations. Fish availability is also calculated through the use of step functions related to pH. Radionuclide concentrations from this model are input into man's diet directly through fish, and indirectly through moose consuming aquatic macrophytes to allow dose calculation for man.

12.2 Comparison of model with requirements

The model uses a conventional approach (Blaylock 1982; CSA 1984) with appropriate choices of parameters. This approach does not consider radionuclide uptake from sediments. Additional justification for this simple approach is provided through reference to more detailed models that produce similar conclusions. The incorporation of a pH function, and the use of lognormal distributions for bioaccumulation factors specific to game fish in northern Canadian waters (low alkalinity, low hardness) is appropriate. One deficiency is the absence of a pathway representing wild rice consumption by man. This pathway may be equivalent in magnitude to the moose pathway, and is particularly relevant for typical northern communities. Given the limited data base, radionuclide concentrations in wild rice could be the same as those in aquatic macrophytes.

12.3 Specific comments on the model

Given the general nature of the research data on the effects of pH on fish populations, the choice of values for the pH step functions is reasonable. The step functions should, however, include an estimate of availability at high pH, provided such values are relevant. Swanson (1983; 1985) has recorded high values close to tailings piles; these areas may be used occasionally by fish on a local basis.

As the model depends almost entirely on the choice of values for the bioaccumulation factors, some justification is required for the use of values based on data taken under conditions of low alkalinity and low hardness. Are these values truly representative of the conditions modelled in the watershed, or are they simply representative of northern Canadian waters? Waters with low mineral contents have high B_f values for uranium and radium (Blaylock 1982). Similarly, bioaccumulation of lead may be enhanced in acid lakes due to the strong pH dependence of lead speciation (Stokes et al. 1985). These factors cannot be specifically modelled at present due to a lack of good data. However, the existing model should clearly specify the range of conditions for which the B_f values are valid, and discuss effects of assumptions on dose predictions.

12.4 Specific comments on parameters

- a) $[F_i]_{n,t}$ is incorrectly specified as to units in section 12.3.2. Correct units are used in Table 12.5.
- b) B_f values appear to have been tabulated from a combination of single points based on one fish, and multiple averages based on many fish. Within studies, the average of values based on many fish sampled under similar conditions should be used as a single data point. Values based on single fish may involve a large analytical error and, therefore, exaggerate the true degree of variation.
- c) Bioaccumulation factors should be updated using the data from Swanson (1985).

12.5 Recommendations

- a. Tabulate bioaccumulation factors based on individual studies rather than specific data points, and recalculate geometric means and geometric standard deviations. Specify range of water conditions for which these B_f values apply.
- b. Incorporate a wild rice pathway to man or justify its exclusion based on dose estimates.

12.6 Summary

- a. The models meet the requirement of estimating radionuclide concentrations in the two aquatic food types by following accepted, conventional procedures.
- b. A narrower range of values could be generated by retabulating existing and additional data for B_f and specifying an expected range of water conditions.

12.7 References

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13. Review of Animal Produce Models (S. Mihok and R. Zach)

13.1 Overview and description of requirements of component model

The animal produce model is required to estimate the concentration of radionuclides in caribou and moose flesh, based on air- and water-related pathways. Radionuclide concentrations are averaged annually over the entire area modelled; flesh concentrations are, therefore, constant over locations and receptors, but can vary with time. Radionuclides enter animal flesh through the drinking of water, and the ingestion of lichens and summer forage (caribou), or browse and aquatic macrophytes (moose). Flesh concentrations, taking account man's diet, are then used to calculate doses to man.

13.2 Comparison of model with requirements

The model uses a conventional approach based on pathways commonly incorporated in agricultural produce models (Baes et al. 1984; CSA 1984). The main requirement of obtaining estimates of radionuclide flesh concentrations of caribou and moose is therefore met by the model. However, these values are likely only accurate to within about an order of magnitude, given the necessity of estimating parameter values for these wild game species. This level of accuracy is acceptable in these assessment food-chain models. The food habits of both species are modelled with only a minimum of detail, presumably due to the lack of better information. A few potentially significant pathways are also missing from the model. The reasons for ignoring these pathways may have to be explained.

13.3 Specific comments on model

The assumptions about the ecology of the moose and caribou are reasonable and necessary to avoid overestimation of values. The method of calculating uptake through transfer coefficients is standard practice and requires no special justification (Baes et al. 1984).

There are two possible deficiencies in the model resulting from the absence of certain pathways. First, there is no inhalation pathway for animals and the potential significance of this pathway is not mentioned. Zach (1985) showed that this pathway can be important for some radionuclides. However, due to lack of data, it would be difficult to implement for caribou and moose. Perhaps, it should at least be mentioned, and its effects on dose predictions discussed. In the particular case of a tailings area, high atmospheric dust values during grazing may magnify dose contributions from inhalation.

A second possible deficiency exists in the absence of a soil ingestion pathway, particularly for caribou, which are known to graze close to the ground. This pathway could be of particular significance during the winter period when lichens are consumed. Zach and Mayoh (1984) have shown that soil ingestion can contribute significantly to dose estimates for radionuclides with low soil/plant concentration ratios such as uranium and

thorium. Radionuclides in ingested soil can be assumed to be transferred to flesh to the same degree as those ingested with forage and drinking water.

13.4 Specific comments on parameters

The values chosen for all parameters were reviewed and were found to be reasonable. Some justification for the choice of extreme values in the triangular distributions is required. In the case of transfer coefficients, the choice of extreme values should be based on the best data available for beef cattle (Baes et al. 1984). Referencing U.S. NRC (1982) is inadequate in this case because there is a more recent review. The source and logic in deriving transfer coefficients should be stated explicitly. Although there are many possibilities, one reasonable approach for transfer coefficients is to specify lognormal distributions with a geometric standard deviation of about 3.0 (see Little 1979 and Ng et al. 1982). Lognormal distributions were also used to quantify radionuclide transfer to aquatic organisms.

13.5 Recommendations

- a. Discuss animals' inhalation pathway and its effect on dose predictions.
- b. Incorporate the soil ingestion pathway for caribou winter feeding on lichens equal to at least 4% soil ingestion by weight of dry lichens (Zach and Mayoh 1984).
- c. Reference the source of information for transfer coefficients explicitly and justify source of values, and degree of variation, taking into consideration the review of Baes et al. (1984).

13.6 Summary

- a. The model meets the requirement for estimating radionuclide concentrations in caribou and moose flesh by extending accepted, conventional procedures to these game species. Effects of excluding animals' inhalation and soil ingestion pathways should be evaluated. Greater latitude in the choice of values for some parameters is desirable to ensure conservative dose estimates.

13.7 References

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14. Review of Dose Models (S. Mihok and R. Zach)

14.1 Overview and description of requirements of component model

The ingestion and inhalation dose model calculates the dose to individuals and populations given various options specifying the location and type of receptor individual. Both critical and average individuals can be modelled with separate options for various parameters. Exposure occurs through ingestion of fish, caribou, moose, vegetables, and drinking water, and through inhalation of air and through groundshine (Section 8). Various assumptions are made about the source of local foods; hence consumption of local beef and milk are not modelled. Immersion in water and exposure to local building materials are also not considered.

14.2 Comparison of model with requirements

The basic goal of estimating dose to man is met by the model structure. Appropriate pathways have been included, and methods recommended by ICRP (1977) and followed by other groups have been adopted. Specific deficiencies lie more in the choice of parameter values than in the model structure itself. In particular, the methods of parameter selection for the critical group are not well presented and documented. In some cases, low parameter values resulting in low doses have been chosen over higher values. This is nonconservative. ICRP-26 methodologies should be thoroughly explained to show that doses from different radionuclides and pathways are fully additive. When such methodologies are not followed, as in the case of thorium, it should be made clear that procedures used give conservative dose estimates.

14.3 Specific comments on the model

Overall, the concept of the critical group is not well defined. If the critical group represents local individuals living off the land (e.g. aboriginal peoples), then the lifestyle of these individuals should be taken into account in the dose calculations. Otherwise one could, perhaps, model extreme versus average individuals.

The critical group seems to be a compromise between concepts of critical versus normal, and extreme versus average. Therefore, the logic in the choice of parameter values is often difficult to follow. Where a choice is available (e.g. dose conversion factors for thorium and radon), low, non-conservative values are sometimes chosen. This may conflict with the concept of a critical group, that is a group consisting of the most highly exposed individuals. If one is to contrast the results of extreme and average groups, it should be done through the use of a larger contrast in parameter values, or through the use of stochastic models. Attempts are made to do this in some cases (e.g. the critical group eats 100% local caribou and moose), but the process is not consistent. In summary, the critical groups should be defined explicitly, and parameter values should be chosen to reflect any assumptions made about the nature of this group.

14.4 Specific comments on parameters

It would be preferable to choose conservative values for parameters where a distinct choice is available from the expert scientific literature. For example, the model uses a value other than the one recommended by ICRP (1981), based on the belief that occupational breathing rates are unrealistically high for members of the public. Although reasonable, this assumption leads to the choice of a dose conversion factor an order of magnitude lower from that recommended by ICRP (1981). This may leave the model open to criticism. Hence, a conservative choice may be preferable.

Similarly, the dose conversion factors for thorium are based on lower values from ICRP (1981) than those recently suggested by J.R. Johnson (personal communication). As thorium uptake is poorly understood (ICRP-48 1986), we suggest that Johnson's values are preferable to maintain a generally conservative approach in the model. Updated values from Johnson ($\mu\text{ Sv} \cdot \text{Bq}^{-1}$) are:

	Ingestion	Inhalation
Th-232	1.9	4.1×10^2
Th-230	3.5	8.1×10^1
Th-228	2.5	1.1×10^2

It should be made clear how daughters were dealt with in the dose calculations. In particular, do the dose-conversion factors include effects of radioactive daughters? If so, which daughters, and were they included by assuming secular equilibrium with the parents? The parameter R_v should be dropped as it is rarely used in contemporary models (Hoffman et al. 1984).

The choice of ingestion rates for vegetables based on both Canadian and U.S. data for different receptors is not explained. The consumption rates are taken from U.S. NRC (1982) and are in error in the original document. The values taken from an unpublished source (Nutrition Canada) are similar to those recommended in CSA (1984), and should be used for both receptors ($157 \text{ kg} \cdot \text{a}^{-1}$ above-ground vegetables, $70 \text{ kg} \cdot \text{a}^{-1}$ below-ground vegetables). A similar justification is required for the choice of different data bases for the consumption of animal meats. Why was a uniform distribution chosen here, in contrast to the triangular distributions chosen nearly everywhere else?

The consumption of fish is modelled without any large extremes, in contrast to the extremes allowed for the consumption of animal meats. This tends to confound the interpretation of the critical group, which might also eat a modest number of fish under certain circumstances.

For inhalation and ingestion of water, the critical receptor spends an average of 60% of his time in the area. Why is an occupancy factor used here and not elsewhere? Similarly, the use of such a low mean occupancy value greatly affects the contrast between critical and average receptors. This again is contrary to the requirements of the model to identify high exposures in a critical group.

14.5 Recommendations

- a. The goals of the model should be more clearly expressed so that the concept of the critical group is clear. If appropriate data are not available for the lifestyle of the critical group, conservative values should be chosen for all parameters to reflect extremes.
- b. Inconsistencies in the degree of conservatism, the choice of data bases, the use of occupancy factors, and the treatment of variation should be rectified.

14.6 Summary

- a. The dose models meet the requirements for predicting individual and population doses, by using accepted ICRP-26 methodology. This methodology could be explained more fully to demonstrate additivity of doses.
- b. The choice of the receptor population characteristics should be better explained to ensure that dose estimates are conservative. This may have to involve a better rationalization of parameter values, and a choice of more conservative values.

14.7 References

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15. Review of Approaches to Modelling Episodic Events (B.V. Goodwin)

15.1 Overview and description of requirements of models for episodic events.

Models for episodic events would be required for all such events that could affect the potential impacts of the uranium tailings disposal system*. The different types of episodic events that could be considered are highly dependent on the characteristics of the disposal system, including site-specific characteristics. This consideration should therefore define a phased approach to the treatment of episodic events. One possible approach is as follows:

- (1) Identification of all potentially-significant episodic events. There should be an exhaustive list of all possible episodic events that might be important. The objective is to identify those events that could significantly modify the "consequences" of the disposal of uranium mill tailings, which are modifications to the pH of nearby water courses and the increased risk to man due to exposure to radioactive and non-radioactive contaminants.
- (2) Identification of significant episodic events. A preliminary examination should be carried out to estimate the magnitudes of each event on the consequences. For example, current proposed criteria for disposal of radioactive waste (AECB, 1986) include a risk limit of 10^{-6} a⁻¹. If some episodic events are very rare, then it may be possible to exclude such events from further detailed analysis, based on preliminary estimates of the probabilities and subsequent health impacts.
- (3) Detailed analysis of significant episodic events. A more detailed analysis should then be carried out for all episodic events that have been identified as significant. This analysis may involve development of suitable models for UTAP. Note, however, that low probability events are best treated as separate cases from each other and from the "reference" case (OECD, 1984).

This phased approach would focus detailed research effort on those episodic events that require further attention. It would also tend to identify those events that should be examined using a variation of the models in UTAP, and those that could be examined using other analytical tools.

* Note that different episodic events could also be regarded as different "scenarios". The OECD (1984) defines scenarios, and presents an outline of the analysis of scenarios and their (possible) treatment with codes such as UTAP.

15.2 Comparison of model with requirements

Appendix C contains a brief review of some naturally-occurring episodic events that have the potential to affect the performance of uranium mill tailings. The events considered are:

- major precipitation/flooding;
- earthquakes;
- very high winds (tornadoes and hurricanes);
- droughts; and
- forest fires.

This list is not exhaustive; for example other potential episodic events would include slumping, dam failure (other than by flooding), glaciation, and creep of the tailings slopes due to cyclic freeze - thaw processes and/or seismic processes. The discussion in Appendix C also explicitly ignores human-caused events, such as removal of tailings material and/or intrusion. This potential "episodic" event may be very serious and warrants special consideration (see, for example, UMT Panel, 1986).

A more orderly and systematic approach would be required to list all potentially-significant episodic events. It may be, however, that this topic was considered to be outside the scope of Appendix C.

The discussion in Appendix C continues with a further examination of the five listed episodic events. The objective was to discuss how these events might be modelled, and to provide specific and/or conceptual recommendations for modifications to the phase III models in UTAP. No models are developed in any detail.

The discussion in Appendix C implies that several of the phase III component models would require modifications to simulate episodic events. In general, all components may require modification. A forest fire or flooding, for example, could cause significant changes to the component models for water source, air source, soil, groundshine, terrestrial vegetation, subsurface water transport, aquatic biota, pH, animal produce and dose.

To simplify discussion, and as a first approximation, Appendix C describes the proposed modelling modifications relative to two categories. These two categories, based upon the immediate consequences of (several of the) episodic events, are as follows:

- bulk loss of tailings and/or tailings porewater (due to events such as floods and/or earthquakes); and
- partial or total reduction of tailings vegetation (due to drought, high winds and/or forest fires).

There is also some separate discussion of drought, which appears to be an extension of the 'reduction of vegetation' category.

In the discussion for these episodic categories, there is an implication that only a few modifications might be required. For example, for the 'bulk loss' category, it is suggested that modifications are

required for the erosion model, together with the first node and first lake of the watershed model. In general, these modifications are:

- (1) Not applicable to all possible episodic events. For example, in the 'bulk loss category', flooding and earthquakes are not equivalent; flooding will contaminate the watershed within a very short period of time, whereas earthquakes may not.
- (2) Not sufficient to model all impacts that could occur. For example, a flood could have much more impact than simple erosion; increased water flow throughout the entire watershed area could lead to contaminated flood plains and lower availability of aquatic life, moose and edible vegetation. In addition, high levels of precipitation could lead to significant changes in the moisture content of the tailings, with subsequent alterations to the chemistry of the tailings pile.

The examples given here are for the 'bulk loss' category, but similar remarks apply to 'reduction of vegetation' and/or to the discussion on drought. There are two major conclusions that might be made:

- (1) It is not advisable to treat two or more episodic events together; and
- (2) Most episodic events have the potential to affect many (or even all) of the component models due to synergistic interactions.

15.3 Specific Comments on model.

Appendix C does not contain any detailed models, and specific comments are therefore not possible. In general, however, it should be restated that many or all of the existing phase III models might require modification for each episodic event considered.

15.4 Specific Comments on proposed parameter modifications.

Although there are no parameter values given in Appendix C, it should be noted that all parameters used by each modified component model could also be affected, and a case-by-case examination of the validity of all parameter values would be required. Some parameters used in the reference models may be invalid when used with an episodic event; for example it may not be possible to extrapolate the empirically-derived equation for the effect of moisture content on radon emanation for both flood and drought conditions.

15.5 Recommendations

- a. Prior to the development of models for episodic events, a systematic attempt should be made to identify all events that could have a significant impact on the tailings disposal

system. It is not sufficient to examine particular events in detail, without first demonstrating that all significant events have been taken into consideration.

- b. An examination of all significant episodic events should be made, to determine which events (if any) should be modelled using UTAP. Other analytical tools may be more appropriate.
- c. Component models for each episodic event should be constructed separately from one another, and treated as special and separate scenarios. It may be possible to use modified component models for more than one episodic event, but each case should be modelled separately.
- d. Each episodic event should be carefully examined to insure that all important effects are modelled. Synergistic effects and coupled effects would be taken into consideration. It is not sufficient, for example, to model the impact of flooding of the tailings without also modelling corresponding impacts (of increased water flows) on the watershed.
- e. Human-induced effects should also be specifically examined as a class of episodic events. Over the lifetime of the tailings pile, man may have a much more serious impact on the behaviour of the disposal system than naturally-occurring events.

15.6 Summary

Appendix C provides a brief outline of possible modifications to the Phase III component models. It concludes that "it is quite possible to construct models to predict the effects of relatively rare events", and the process would involve "considerable work, primarily regarding logic flow and internal bookkeeping". It is probably true that episodic events could be modelled and certainly the task is not trivial. It appears, however, that not all aspects of the problem have been examined. Episodic events are likely to result in many significant changes to all or most of the component models, instead of a small number of changes to a few models. The modelling challenge is without doubt much more complex than is implied.

15.7 References

- UMT Panel, 1986. Scientific Basis for Risk Assessment and Management of Uranium Mill Tailings. Uranium Mill Tailings Study Panel, Board on Radioactive Waste Management, National Research Council, 2101 Constitution Avenue NW, Washington D.C., National Academy Press, 1986.
- AECB, 1986. Regulatory Objectives, Requirements and Guidelines for the Disposal of Radioactive Wastes. Atomic Energy Control Board Consultative Document C-104, 1986.
- OECD, 1984. Long-Term Radiation Protection Objectives for Radioactive Waste Disposal. Organization of Economic Cooperation and Development Report No ISBN 92-64-12604-I, OECD Nuclear Energy Agency, Paris, 1984.

16. Summary of review and recommendations

In conclusion, we note that the UTAP model, developed by SENES under contract to NUTP, represents a significant advancement towards the environmental assessment of uranium mill tailings. The UTAP authors have combined a wide range of environmental models, including a relatively new model for the tailings source term. Most of our critical comments have focused on specific details, and should not overshadow the significant assessment framework developed by UTAP. The following summary is a condensed list of the most important points of our review.

General comments

- The model considers a reasonable, but single, scenario. Neither intrusion (man living on tailings) nor large-scale agricultural practices are modelled.
- The documentation is weak for all of the component models, and pathways that were excluded should be discussed. We have suggested some editorial changes to strengthen the documentation.
- We have pointed out inconsistencies and inadequacies in each section. Many of these should likely not greatly affect the predictions of the dose to man.
- We have suggested minor changes to several parameter values in each component model.
- The UTAP model, as a whole, is acceptable for the long-term environmental assessment of uranium mill tailings, for the given scenario.

Uranium mill tailings water source

- The overall modelling concept is reasonable, but we have made some suggestions concerning uranium solubility and the electroneutrality equations.
- Some probability distribution functions (such as those for the K_d values) can be improved.
- It would be desirable to compare the results of the model calculations with field data from a tailings area.

Tailings erosion model

- The model, based on the Universal Soil Loss Equation, is adequate, but floods are likely important and should be considered.
- The parameter values need better justification.

Air source model

- The models are satisfactory, but suspension by mechanisms such as fire should be addressed.
- Suggestions for improvement of some parameter distributions are given.

Air dispersion model

- The model, with some changes, calculates air concentrations satisfactorily for distances >500 m from the tailings source. The model does not adequately calculate air concentration over the tailings itself.

Soil model

- The model misses some potentially important pathways and would likely underestimate the dose to man.
- We recommend the inclusion of upward movement of radionuclides from contaminated groundwater into the soil, and contamination through surface irrigation of gardens.
- Sensitivity analyses and tests of the model with interpretations are needed.

Groundshine model

- This model is adequate to predict external gamma dose rate for long-term assessment of uranium tailings.
- A discussion of beta doses is required.

Terrestrial vegetation model

- The model is generally satisfactory but the calculation of radionuclide distribution and concentration in the root zone should include the litter layer.
- Missing pathways should be addressed. These include transport of radionuclides from aquatic sediments to vegetation, and irrigation during gardening.
- We have recommended changes to some of the parameter values.

Subsurface transport models

- The general concept and implementation of the model is satisfactory.
- There are inconsistencies in some of the calculations, and inadequate consideration of migration of radionuclide chains and ingrowth of daughter nuclides.

Surface water dispersion models

- The model meets most of the requirements of the assessment.
- We have recommended changes to some parameter values, and the model should be verified using available field data.

Aquatic models

- The model meets the requirement for estimating radionuclide concentrations in the two specified food types.

- The parameter values require improvement.
- Ingestion of aquatic vegetation, such as wild rice, should be included.

Animal produce models

- The basic model is satisfactory but the exclusion of some pathways, such as animal inhalation and soil ingestion, should be addressed.
- Some parameter values have not been selected conservatively, and the model may underestimate the dose to man.

Dose models

- The model meets the basic requirements of predicting dose to man through various pathways.
- The receptor and population characteristics need better definition. The parameter values for the critical group need better justification and are not always conservative. This may cause underestimation of the dose to man.

Episodic events

- We have provided some suggestions and criteria for selection of episodic events if these are incorporated into the UTAP code. The inclusion of episodic events is likely to affect all of the component models, perhaps significantly.

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